OD3.3

Natural habitats for coastal protection and relevant multi-stressor coastal risks. Report and European scale overview

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CONTENTS

I. GENERAL INTRODUCTION

Within the context of THESEUS, ecosystems are considered in 3 different functions/roles:

- Receptors of flooding, which may either cause loss or shift of existing ecosystems and in some cases the creation of new ecosystems.
- Pathways via which water flooding can move to other regions. In being a pathway, the ecosystem may affect flooding risk (e.g. some ecosystems can attenuate waves).
- Mitigation options that may be created to contribute to coastal defense systems.

Within this report, we will regard 3 dominant coastal ecosystems that form striking features in the landscape and are known or hypothesized, to play an important role in coastal defense: coastal dunes, salt marshes and biogenic reefs (e.g. oyster reefs, mussel beds, etc.). For each of these ecosystems, we will:

i) Briefly present where they can be found across Europe (based on ID 3.1) which is important from the receptor and pathway perspective.

ii) Provide an overview of what is known about their coastal defense values (based on text written for WT2.2) and which measurements are carried out within THESEUS (ongoing and finalized) to extent on this knowledge. This is important for the receptor and mitigation perspective.

iii) Provide a mechanistic insight into how these ecosystems function, use this to explain which risks they are facing and how this knowledge is used (or not) in current management (based on text written for WT1.4, WT1.5, WT2.2 and ID3.1). This knowledge is important from the receptor and mitigation perspective.

iv) Briefly summarize the work performed within THESEUS (ongoing and finalized) to improve management practices and opportunities for restoring and creating these ecosystems, which is important from a mitigation perspective.

Table 1.1. Overview of chapters and their relevance to the THESEUS view on ecosystems as Receptors, Pathways and Mitigation options.

Chapter	Title	Receptor	Pathway	Mitigation
2	Distribution of dunes, salt marshes & biogenic reefs that may contribute to coastal defense: an illustration derived from the GIS database	**X**	**X**	
3	Coastal protection value of dunes, salt marshes & biogenic reefs: a literature review and overview of ongoing and finished measurements		**X**	**X**
4	Mechanistic description of dunes, salt marshes & biogenic reefs: a general explanation of the natural ecosystem dynamics, threats and how this knowledge is relevant to management	**X**		**X**
5	Overview of experiments (and some preliminary results) on how to improve current management practices and possibilities to restore or create these ecosystems			**X**

II. DISTRIBUTION OF DUNES, SALT MARSHES & BIOGENIC REEFS THAT MAY CONTRIBUTE TO COASTAL DEFENSE: AN ILLUSTRATION DERIVED FROM THE GIS DATABASE[1]

II.1. DUNES

[Based on text written for ID3.1]

II.1.1. EUROPEAN-SCALE DISTRIBUTION OF DUNES

European territory has seven regional seas with very different biodiversity and pressures. Coastal dunes are widely distributed around the European coast except for rocky coastlines and in the Boreal region. A map of the distribution of coastal dunes has been prepared within THESEUS based on an analysis of aerial photography (Fig.2.1).

These European dunes show a great diversity of morphological and botanical aspects. Five major regions can be distinguished (Heslenfeld *et al.*, 2004):

- The Baltic region: with small and isolated dunes on the Scandinavian coasts, and bigger systems in the southern part.
- The North Sea region (Northern France, Belgium, the Netherlands, Germany, Denmark) with wide dunes on barrier islands, coastlines and estuaries.
- The Atlantic region: Ireland and UK showing small dune areas; and France, Portugal and Spain showing extensive dune fields.
- The Mediterranean region where larger dune fields are related to deltas and coastal plains.
- The Black sea region with wide-spread dunes along the north western coast.

Figure 2.2 illustrates the succession of natural habitats, from the beach to the rear of the dune which is typical in dunes of the Atlantic, North Sea and Baltic coasts,

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Figure 2.1. Distribution of coastal dunes

Countries	Coastal dunes (km)
Albanie	268,9
Allemagne	454,8
Belgique	59,59
Bulgarie	97,84
Croatie	6,4
Danemark	937,9
Espagne	1077
Estonie	461,7
Finlande	113,8
France	1355
Grande Bretagne	2004
Grèce	833,8
Irlande	807,2
Islande	407,5
Italie	1409
Lettonie	514,9
Lituanie	93,15
Norvège	56,54
Pays Bas	462,2
Pologne	442,1
Portugal	691,9
Roumanie	165,1
Serbie et Montenegro	13,76
Suède	500,9
TOTAL	13234,98

Table 2.1.Coastal dune length in European countries

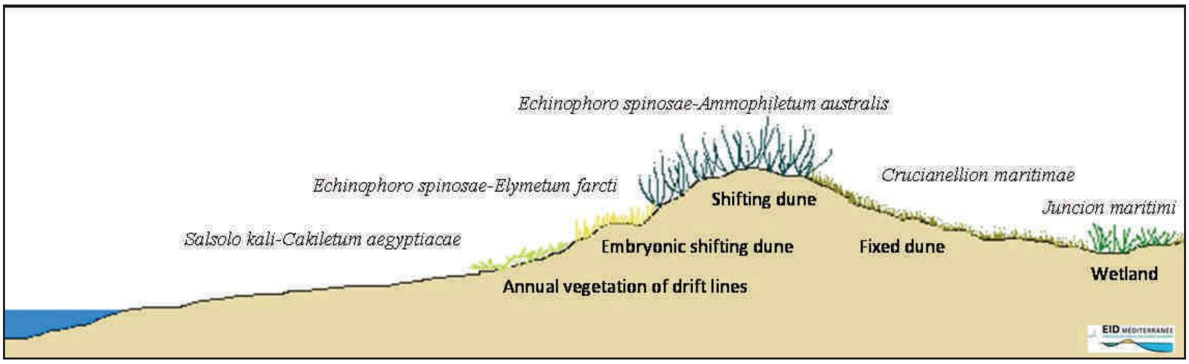


Figure 2.1.: Dune section showing the succession of natural habitat

Natural habitats are described:

- **Annual vegetation of drift lines 1210, PAL.CLASS.: 17.2**

- **Embryonic shifting dunes 2110, PAL.CLASS.: 16.211**

- **Shifting dunes along the shoreline with *Ammophila arenaria* (white dunes) 2120, PAL.CLASS.: 16.212**

Regional variations can be found in shifting dunes, implying different vegetal associations. The main associations are presented in the following table:

Shifting dune variations	
PAL.CLASS	**Vegetal associations**
16.2121	*Ammophila arenaria*
	Eryngium maritimum
	Euphorbia paralias
	Calystegia soldanella
	Otanthus maritimus
	Elymus arenarius
16.2122	*Ammophila arenaria*
	Medicago marina
	Anthemis maritima
	Cutandia maritima
	Euphorbia paralias
	Eryngium maritimum
	Echinophora spinosa
16.2123	*Zygophyllum fontanesii*
	Polygonum maritimum
	Ononis natrix
	Cyperus capitatus
	Polycarpaea nivea
	Euphorbia paralias
	Convolvulus caput-medusae
	Androcymbium psammophilum

The same way, different vegetal association of fixed coastal dunes with herbaceous vegetation (grey dunes) can be found in European coasts:

Fixed dune variations	
PAL.CLASS	**Vegetal associations**
16.221	Norhtern grey dunes with grass communities and vegetation from ***Koelerion albescentis, Corynephorion canescentis, Sileno conicae-Cerastion semidecandri***
16.222	Biscay grey dunes (***Euphorbio-Helichrysion stoechadis***): on stabilised humus soil infiltrated by dwarf bushes with *Helichrysum stoechas, Artelisia campestris* and *Ephedra distachya*
16.225	Atlantic dune (*Mesobromion*) grasslands: various sandy coastal sites characterised by herbaceous vegetation in the form of calcicol mesoxerocline grasslands, corresponding to the communities of *Mesobromion* found by the sea; dunal grasslands composed of species characteristic of dry calcareous grasslands
16.226	***Trifolio-Geranietea sanguinei: Galio maritimi-Geranion sanguinei, Geranium sanguineum*** formations on neutro basic soils rich in calcium and poor in nitrogen
16.227	Dune fine-grass annual communities: sparse pioneer formations of fine grasses rich in spring-blooming therophytes characteristic of oligotrophic soils

· **Decalcified fixed dunes with *Empetrum nigrum* PAL.CLASS.: 16.23, 2140**

Dunes colonised by *Empetrum nigrum* heaths of the coasts.

· **Atlantic decalcified fixed dunes (*Calluno-Ulicetea*) PAL.CLASS.: 16.24, 2150**

Decalcified dunes of France, Belgium and Britain, colonised by heaths of the alliances *Calluno-Genistion* or *Ulicion minoris*, and of Iberia, colonised by heaths of the alliance *Ericion umbellatae*.

· **Dunes with *Hippophae rhamnoides* PAL.CLASS.: 16.251, 2160**

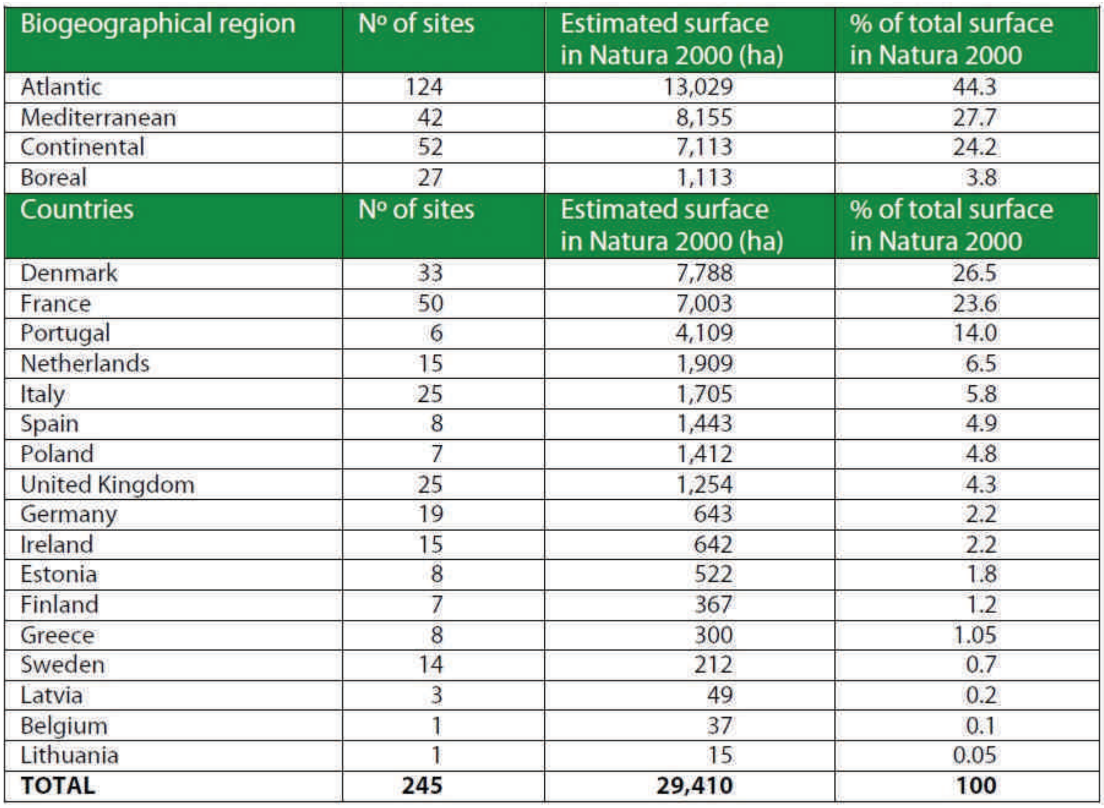
· **Dunes with *Salix repens* ssp. *argentea* (*Salicion arenariea*) PAL.CLASS.: 16.26, 2170**

Salix repens communities (*Salicion arenariae*), colonising wet dune slacks. Following the lowering of the ground water table or accumulation of drift sand

· **Wooded dunes of the Atlantic, Continental and Boreal region PAL.CLASS.: 16.29, 2180 (**Error! Reference source not found.**)**

· **Humid dune slacks PAL.CLASS.: 16.3 = 16.31 to 16.35, 2190**

Figure 2.2. Wooded dunes



Biogeographical region	Nº of sites	Estimated surface in Natura 2000 (ha)	% of total surface in Natura 2000
Atlantic	124	13,029	44.3
Mediterranean	42	8,155	27.7
Continental	52	7,113	24.2
Boreal	27	1,113	3.8
Countries	Nº of sites	Estimated surface in Natura 2000 (ha)	% of total surface in Natura 2000
Denmark	33	7,788	26.5
France	50	7,003	23.6
Portugal	6	4,109	14.0
Netherlands	15	1,909	6.5
Italy	25	1,705	5.8
Spain	8	1,443	4.9
Poland	7	1,412	4.8
United Kingdom	25	1,254	4.3
Germany	19	643	2.2
Ireland	15	642	2.2
Estonia	8	522	1.8
Finland	7	367	1.2
Greece	8	300	1.05
Sweden	14	212	0.7
Latvia	3	49	0.2
Belgium	1	37	0.1
Lithuania	1	15	0.05
TOTAL	**245**	**29,410**	**100**

Table 2.1 : Estimation of humid dunes slacks surfaces (habitat surface included in Natura 2000)

Humid dune slacks	
PAL.CLASS	Vegetal associations
16.31	Dune-slack pools (*Charetum tomentosae, Elodeetum canadense, Hippuridetum vulgaris, Hottonietum palustris, and Potametum pectinati*): fresh-water aquatic communities of permanent dune-slack water bodies
16.32	Dune-slack pioneer swards (*Juncenion bufonii* p.: *Gentiano-Erythraeetum littoralis, Hydrocotylo-Baldellion*): pioneer formations of humid sands and dune pool fringes, on soils with low salinity.
16.33	Dune-slack fens: calcareous and, occasionally, acidic fen formations, often invaded by creeping willow, occupying the wettest parts of dune-slacks.
16.34	Dune-slack grasslands: humid grasslands and rushbeds of dune-slacks, also often with creeping willows (*Salix rosmarinifolia* , *S. arenaria*),
16.35	Dune-slack reedbeds, sedgebeds and canebeds: reedbeds, tall-sedge communities and canebeds of dune-slacks

· **Machairs PAL.CLASS.: 1A, 21A0**

Complex habitat comprised of a sandy coastal plain resulting partially from grazing and/or rotational cultivation, in an oceanic location with a cool, moist climate.

From the beach to the rear of the dune we can find on the sea dunes of dunes of the Mediterranean coast:

· ***Crucianellion maritimae* fixed beach dunes PAL.CLASS.: 16.223, 2210**

Fixed dunes of the western and central Mediterranean (**Error! Reference source not found.**), of the Adriatic, of the Ionian Sea and North Africa with *Crucianella maritima*, *Pancratium maritimum*.

Figure 2.3. **Crucianellion maritimae fixed dunes**

· **Dunes with *Euphorbia terracina* PAL.CLASS.: 16.224, 2220**

Coastal dune grassland communities, with, among others, *Euphorbia terracina* (**Error! Reference source not found.**), *Silene nicaeensis, Ephedra distachya* and *Silene subconica*.

Figure 2.4. *Euphorbia terracina*

· ***Malcolmietalia* dune grasslands PAL.CLASS.: 16.228, 2230**

· ***Brachypodietalia* dune grasslands with annuals PAL.CLASS.: 16.229, 2240**

Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea: Meso- and thermo-Mediterranean xerophile, mostly open, short-grass perennial grasslands rich in therophytes; therophyte communities of oligotrophic soils on base-rich, often calcareous substrates.

· **Coastal dunes with *Juniperus* spp. PAL.CLASS.: 16.27 and 64.613, 2250 (**Error! Reference source not found.**)**

Figure 2.5. Wooded dunes with *Juniperus sp.*

· ***Cisto-Lavenduletalia* dune sclerophyllous scrubs PAL.CLASS.: 16.28, 2260**

Sclerophyllous or lauriphyllus scrubs established on dunes of the Mediterranean and Warm-Temperate Humid regions.

· **Wooded dunes with *Pinus pinea* and/or *Pinus pinaster* PAL.CLASS.: 16.29 x 42.8, 2270**

· **Boreal Baltic sandy beaches with perennial vegetation PAL.CLASS.: 16.1222, 16.132, 16.133, 16.2133, 1640**

Sheltered to exposed, gently sloping sand beaches influenced by wave action, giving a high representation of perennial plant species. The vegetation is often sparse and large areas of bare sand are common especially in the part closest to the shore. Sand-binding plants are common: *Ammophila arenaria, Lathyrus japonicus* ssp. *maritimus, Leymus arenarius, Atriplex littoralis, Salsola kali, Crambe maritima, Honkenya peploides, Cakile maritima, Elytrigia juncea ssp. boreoatlantica.*

	Strandline	**Embryonic/Foredune**	**Fixed dunes**	**Dune slacks**	**Woodunes**
Danemark		*Ammophila arenaria*	*Calluna vulgaris*	*Phragmites australis*	*Quercus robur*
			Carex arenaria	*Scirpus maritimus*	*Picea sitchencis*
			Corynephorus canescens		*Pinus sylvestris*
			Empetrum nigrum		*Pinus mugo*
			Erica tetralix		
Finland	*Cakile maritima*	*Elymus arenarius*	*Deschampsia flexuosa*	*Carex spp*	*Prunus padus*
	Honckenya peploides		*Carex arenaria*	*Phragmites australis*	*Betula spp*
	Lathyrus japonicus		*Empetrum nigrum*	*Salix spp*	*Salix caprea*
			Festuca ovina	*Juncus spp*	*Picea abies*
			Festuca rubra		*Pinus sylvestris*
			Juncus balticus		*Sorbus aucuparia*
			Juniperus communis		
			Myrica gale		
			Salix phylicifolia		
			Salix repens		
France	*Cakile maritima*	*Elytrigia juncea*	*Helichrysum stoechas*	*Schoenus nigricans*	*Quercus ilex*
(Atlantic coast)	*Atriplex arenaria*	*Ammophila arenaria*	*Festuca arenaria*	*Spiranthes aestivalis*	*Quercus suber*
	Salsola kali	*Elymus arenarius.*	*Corynephorus canescens*	*Liparis loeselii*	*Pinus pinaster*
		Eryngium maritimum		*Serpia lingua*	
		Euphorbia paralias			
		Calystegia soldanella			
Great Britain	*Cakile maritima*	*Elytrigia juncea*	*Vulpia membranacea*	*Parnassia palustris*	
	Atriplex arenaria	*Ammophila arenaria*	*Hordeum marinum*	*Liparis loeselii*	
	Salsola kali	*Calystegia soldanella*	*Vulpia ambigua*	*Pyrola rotundifolia*	
	Mertensia maritima	*Eryngium maritimum*	*Calluna vulgaris*	*Epipactis palustris*	
	Lathyrus japonicus		*Erica cinerea*		
			Trifolium subterraneum		
			Trifolium glomeratum		
			Trifolium ornithopodioides		
Iceland	*Cakile arctica*	*Elymus arenarius.*	*Festuca rubra*	*Salix repens*	
	Honckenya peploides	*Mertensia maritima*	*Silene maritima*		
	Atriplex longipes	*Festuca rubra*	*Festuca ovina*		
		Silene maritima	*Equisetum arvensis*		
		Rumex acetosella	*Poa sp*		
		Potentilla anserina	*Salix spp*		
Ireland	*Atriplex laciniata*	*Ammophila arenaria*	*Lotus corniculatus*		
	Cakile maritima	*Elymus arenarius.*	*Viola tricolor*		
	Galium aparine		*Thymus drucei*		
	Elytrigia juncea		*Viola tricolor*		
			Echium vulgare		
			Calluna vulgaris		
			Erica cinerea		
Italy	*Cakile maritima*	*Medicago marina*	*Scabiosa argentiea var. alba*		*Quercus ilex*
		Elytrigia juncea	*Brometalia spp*		*Pinus pinea*
		Eryngium maritimum	*Crucianella maritima*		*Pinus halepensis*
			Juniperus macrocarpa		*Pinus pinaster*
			Pistacia lentiscus		
			Phillyrea angustifolia		
			Daphne gnidium		
Norway	*Atriplex sp*	*Elytrigia juncea ssp. boreali-atlantica.*	*Corynephorus canescens*	*Salix repens*	*Corylus avellana*
	Cakile arctica	*Ammophila arenaria*	*Carex maritima*	*Juncus balticus*	*Quercus robur*
	Honckenya peploides	*Elymus arenarius.*	*Empetrum nigrum*		
			Calluna vulgaris		
Poland	*Salsola kali*	*Ammophila arenaria*	*Corynephorus canescens*		*Quercus robur*
	Honckenya peploides	*Elymus arenarius.*	*Carex arenaria*		*Myrica gale*
	Cakile maritima	*Calammophila baltica*	*Helichrysum arenarium*		*Betula pendula*
			Jasione montana var. littoralis		*Erica tetralix*
			Hieracium umbellatum var dunense		
			Sedum acre		
			Empetrum nigrum		
			Calluna vulgaris		
Portugal	*Cakile maritima*	*Elytrigia juncea ssp. boreali-atlanticus*	*Corema album*		*Pinus pinaster*
	Honckenya peploides	*Eryngium maritimum*	*Stauracanthus gelstoides*		*Juniperus phoenicea*
	Salsola kali	*Euphorbia paralias*	*Thymus carnosus*		*Quercus spp.*
		Elytrigia juncea ssp. farctus	*Crucianella maritima*		
		Ammophila arenaria			
		Othanthus maritimus			
		Otanthus maritimus			
		Calystegia soldanella			
		Artemisia campestris ssp. maritima.			
		Medicago marina			
Spain	*Salsola kali*	*Elytrigia juncea*	*Ononis natrix subsp. ramosissima*		*Juniperus macrocarpa*
	Polygonum maritimum	*Calystegia soldanella*	*Crucianella maritima*		*Juniperus phoenicea*
	Cakile maritima	*Eryngium maritimum*	*Helichrysum stoechas*		*Rhamnus oleoides*
		Sporobolus pungens	*Teucrium polium*		*Phillyrea angustifolia*
		Ammophila arenaria	*Malcolmia littorea*		*Pistacia lentiscus*
		Otanthus maritimus	*Thymus carnosus*		*Tamarix africana*
		Lotus creticus	*Ephedra distachya*		*Corema album*
		Echinophora spinosa			*Olea europaea var. sylvestris*
		Medicago marina			*Quercus coccifera*
					Tamarix gallica
					Carex nigra
Sweden	*Atriplex littoralis*	*Ammocalamagrostis baltica*	*Empetrum nigrum*	*Phragmites australis*	*Pinus sylvestris*
	Honckenya peploides	*Ammophila arenaria*	*Corynephorus canescens*		*Quercus robur*
	Cakile maritima	*Elymus arenarius.*	*Calluna vulgaris*		*Betula pendula*
			Salix arenaria		

Table 2.2 . List of species per habitats around Europe (modified from Doody, 1991 and 2008)

II.1.2. *Examples of temporal variability*

- Santander pilot site

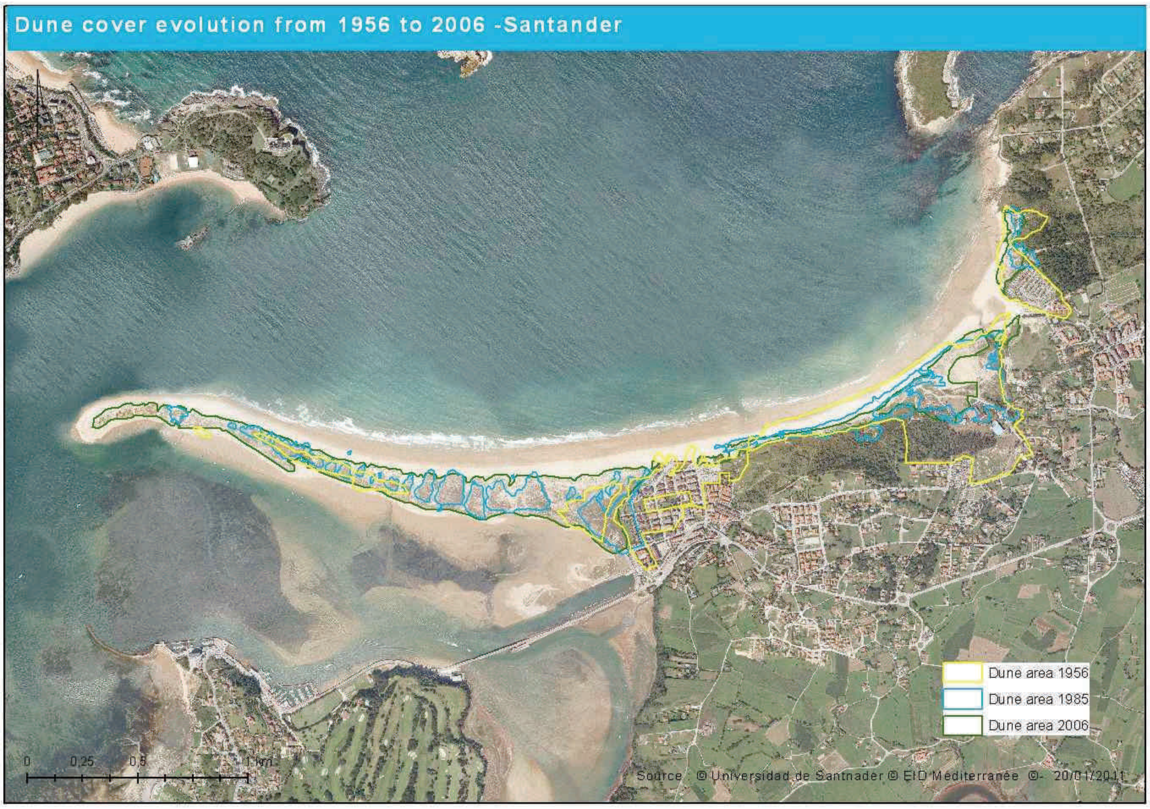
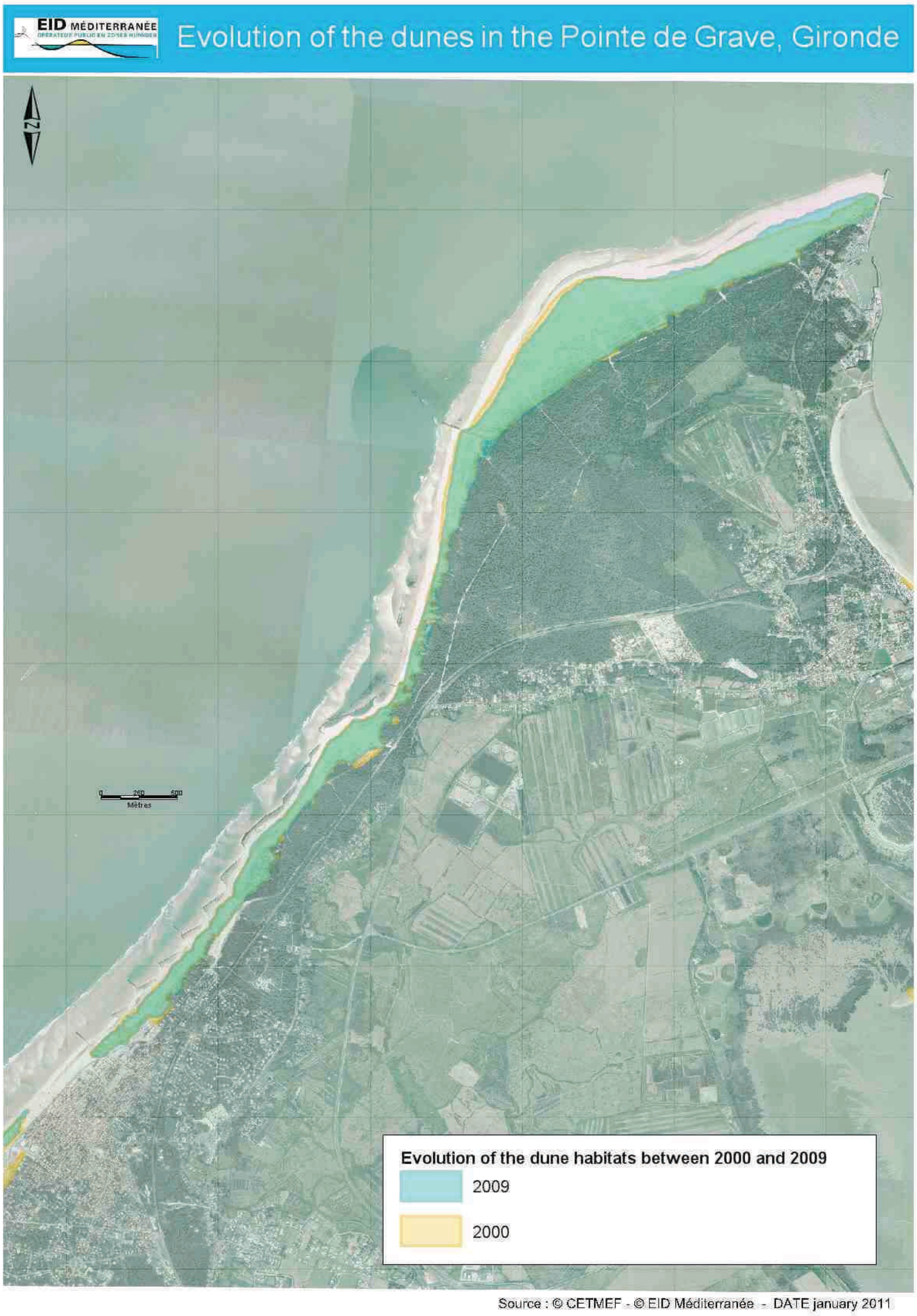


Figure 2.7 . Aerial photograph showing variability in the extent of sand dunes in the Santander site 1956-2006

A general retreat in extent was observed on the east part of the dune system, while a progressive extension is occurring on the western part of the system (Fig.2.7).

- Gironde Estuary

A diachronic photography of the Gironde coastal dunes has been made for 2009 and 2000 (Fig.2.8).



Source : © CETMEF - © EID Méditerranée - DATE january 2011

Figure 2.8. Evolution of the dune system between 2000 and 2009 Gironde estuary spit

The following table shows the area lost between 2000 and 2009.

	2000	2009	Lost between 2000 and 2009
Dunes area	8,39	1,43	6,97

Table 2.4. Dunes area between 2000 and 2009 in km^2

Figure 2.9. Evolution of the dune system between 2000 and 2009 Gironde estuary spit

On the bottom of the estuary spit (Fig.2.9), the coastal dunes are in global retreat.

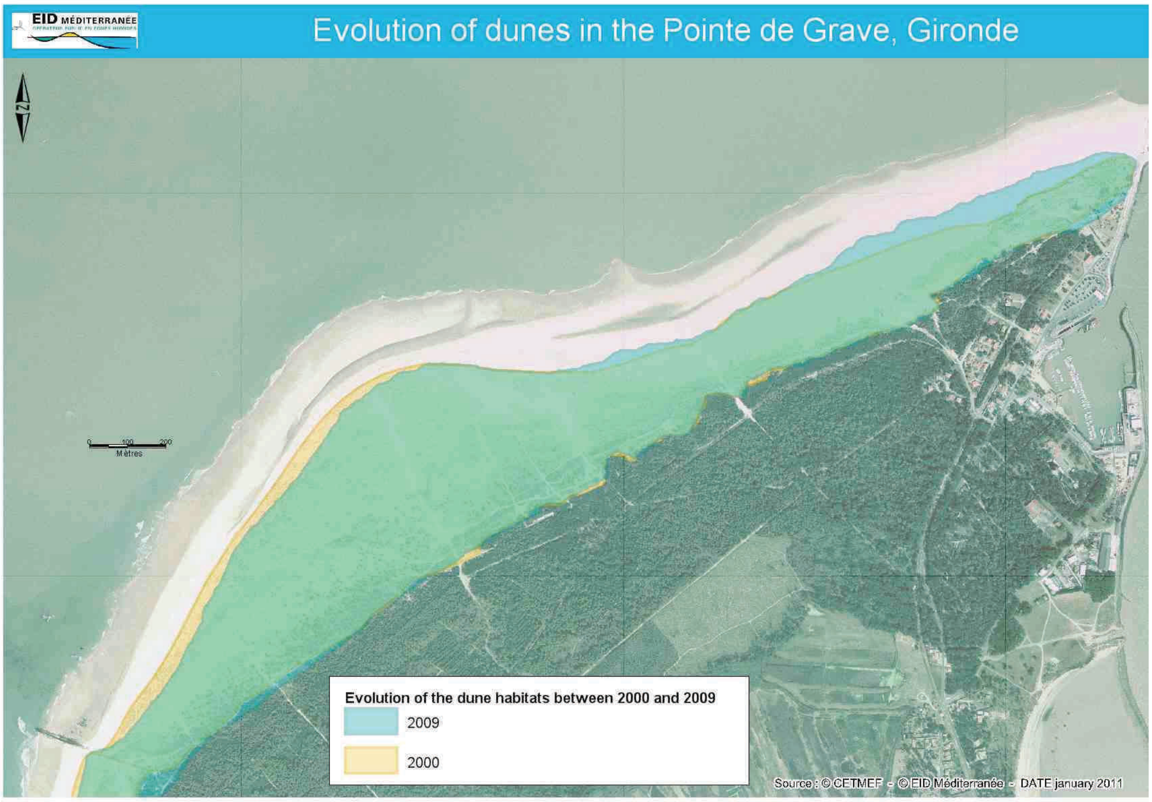


Figure 2.10. Evolution of the dune system between 2000 and 2009 north part of the estuary spit.

While at the northern part of the estuary spit we can see that there is a regressive dynamic of the dune of the south-west portion and a progressive trend on the north-east part (Fig. 2.10).

II.2. SALT MARSHES

[Based on text written for ID3.1]

II.2.1. EUROPEAN-SCALE DISTRIBUTION OF SALT MARSHES

Species composition and zonation of marsh vegetation

Species compositions of vegetation communities are governed by species' ability to compete and their tolerance to site-specific environmental conditions and physical disturbance (Grime 1979). Most classification schemes recognise four vegetation zones in salt-marshes along the vertical gradient in seawater exposure (inundation frequency) (Fig 2.11). To some degree, this vertical zonation reflects a general decline from the low to high elevations in the tolerance of species to seawater flooding (Allen 2000). Thus, the grazing-marsh above the intertidal zone has less predominance of salinity-tolerant species than the salt marsh. However, the predictable vertical alignment of species along the salinity gradient is strongly modified by the species' competitive abilities, as well as their tolerances to site-specific conditions, including tidal range and climate (e.g. rainfall), as well as salinity, nutrient and disturbance (e.g. waves or grazing) regimes (Allen 2000; Bertness et al., 2002). For instance, intense grazing disturbance may cause less competitively strong mid-marsh species to become more common in the high marsh (Kiehl et al. 1996). Traits for competition, environmental and disturbance tolerance often exist in trade-off. Thus, while competitive fitness generally increases up-shore, some low marsh species compensate for lack of competitive ability by being more tolerant to wave disturbance than high shore species (Pennings and Calloway 1992).

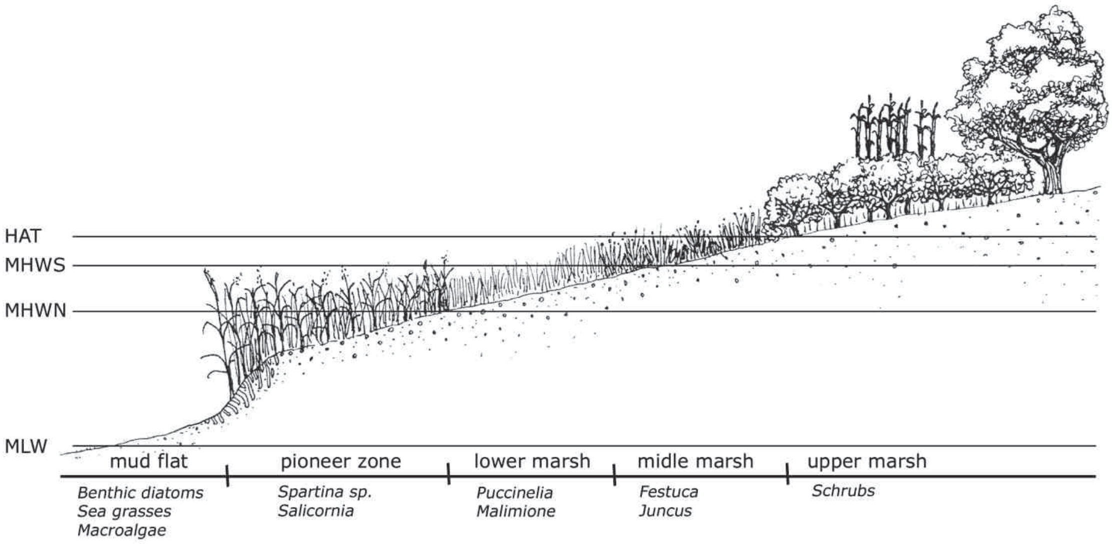


Figure 2.11. Typical salt-marsh zonation (modified from Bertness et al., 2002). Species along the tidal elevation gradient are adapted to the inundation frequency, including extreme flooding and storm events.

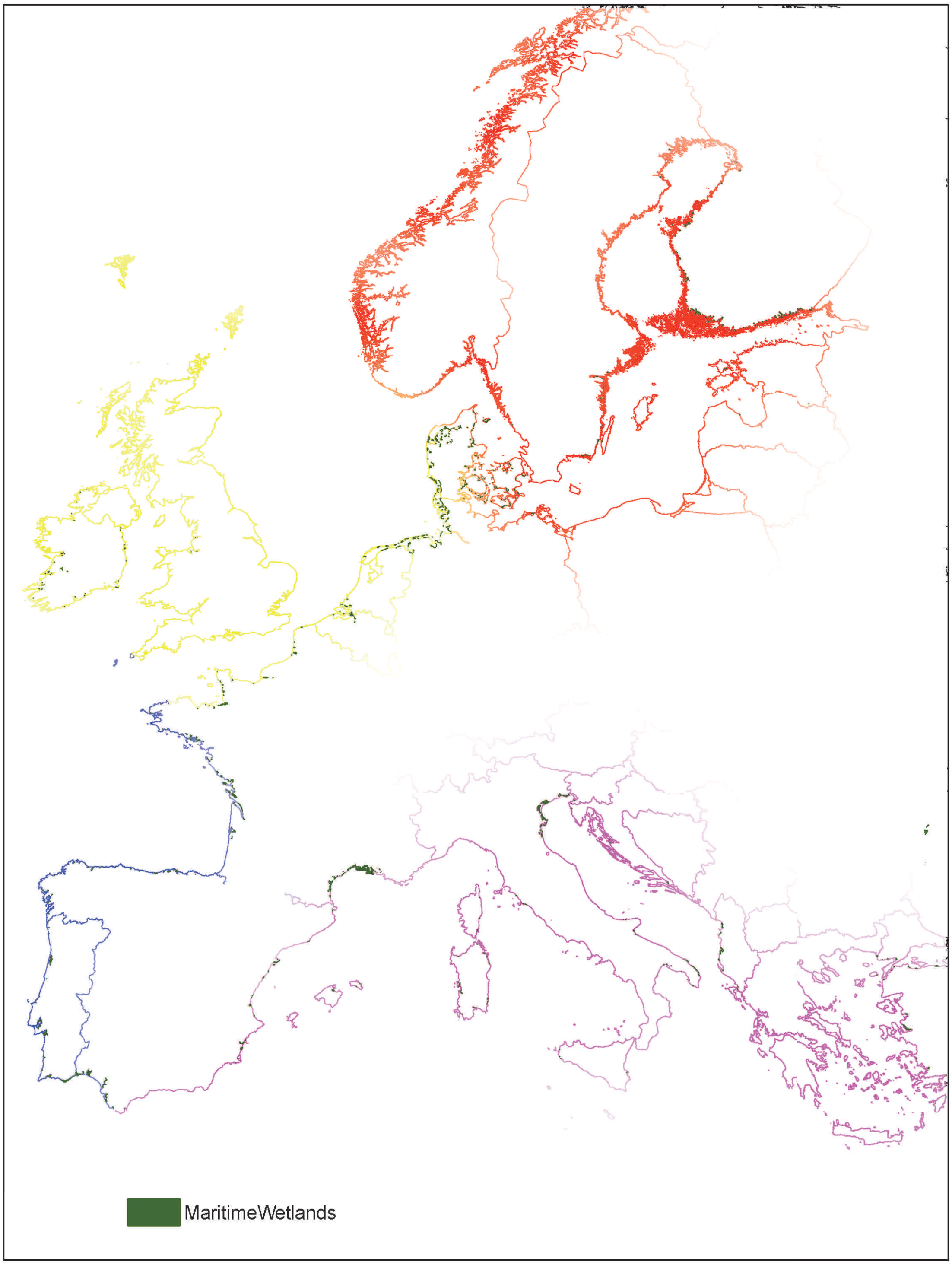


Figure 2.12. Salt marshes across Europe divided to regional type: (red) Baltic and boreal, (yellow) North Atlantic, (blue) south Atlantic, and (purple) Mediterranean.

European salt-marsh distribution and vegetation composition

Although salt-marshes are not the most diverse ecosystems, due to the influence of the salinity regime, they are highly productive. Along the European coasts salt-marsh vegetation composition differs between four regions (Fig. 2.12) from which a large area is protected within Natura 2000 (Fig. 2.13): 1) the North Atlantic; 2) South Atlantic; 3) Mediterranean; 4) Baltic and Boreal regions (European Commission, 2007). A short description of these regional vegetation differences in vegetation composition is given in Table 2.6. In general the salt marshes in North-West Europe can be considered to belong to the North Atlantic. In contrast to the other types, these marshes in most cases are exposed to large tidal amplitudes. In general the salt marshes and meadows along the Baltic Sea experience minimal tidal differences, and can be considered micro-tidal. Most of the Baltic and boreal coastal areas were traditionally used for mowing or grazing, thus enlarging the areas and keeping the vegetation low, rich in vascular plants, characteristically the vegetation occurs in distinct zones, with saline vegetation closest to the sea. The Mediterranean and South Atlantic coastal areas consist largely of the same species composition. This is due to the temperature, although Mediterranean species higher up on the shore are usually more dessication resistant. In general the salt marshes and meadows along the Mediterranean experience minimal tidal differences, and can be considered micro-tidal.

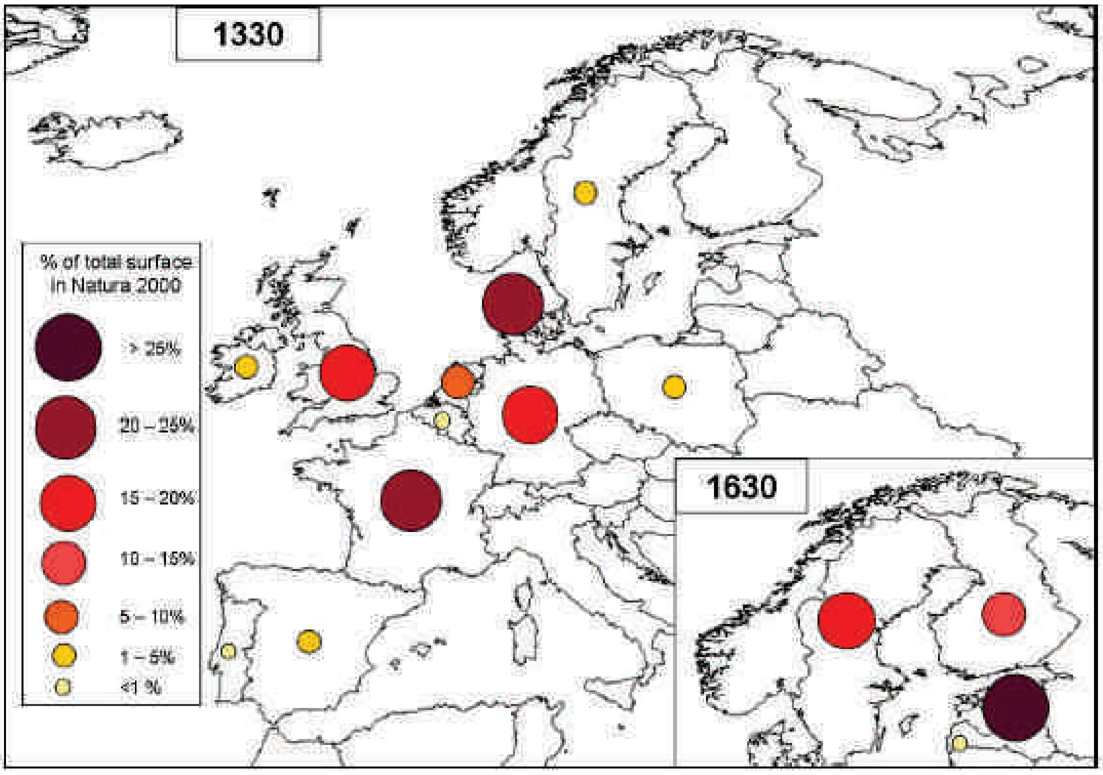


Figure 2.13. Percentage distribution of the total surface of Atlantic salt meadows (1330) and Boreal Baltic coastal meadows (1630) in Natura 2000 (from Doody, 2008).

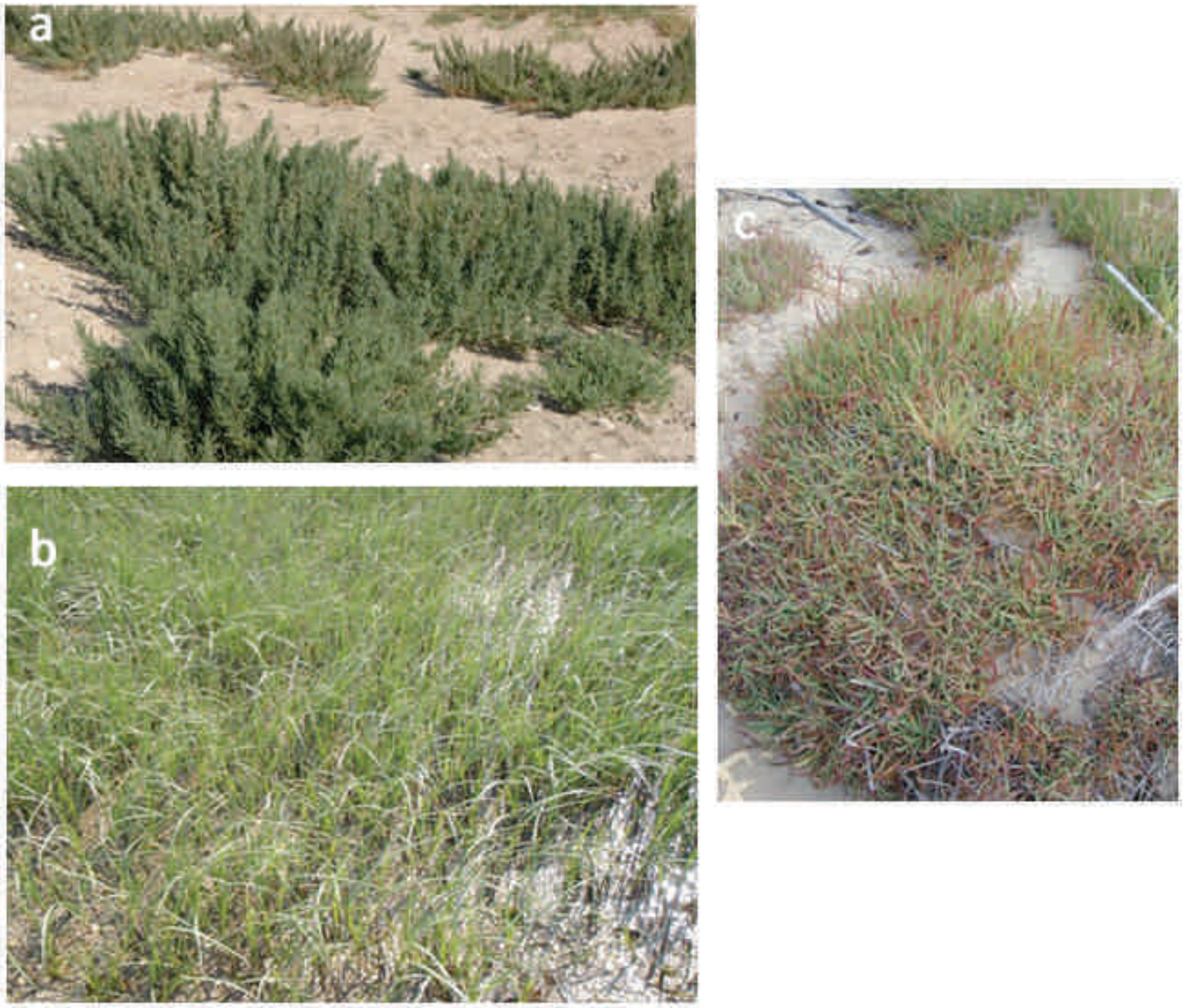


Figure 2.14. Three salt-marsh species: (a) *Suaeda maritima*, (b) *Spartina versicolor,* (c) *Plantago crassifolia*

Table 2.6. European-scale salt-marsh types and subhabitats.

North Atlantic salt-marshes	
Salicornia *spp. and other annuals colonising mud and sand (PAL.CLASS.: 15.1, 1310):*	
	Glasswort swards (Thero-Salicornietalia): annual glasswort (*Salicornia* spp., Microcnemum coralloides), seablite (*Suaeda maritima*), or sometimes saltwort (*Salsola* spp.) formations colonising periodically inundated muds of coastal salt marshes and inland salt-basins.
	Mediterranean halo-nitrophilous pioneer communities (*Frankenion pulverulentae*): formations of halo-nitrophilous annuals (*Frankenia pulverulenta, Suaeda splendens, Salsola soda, Cressa cretica, Parapholis incurva, P. strigosa, Hordeum marinum, Sphenopus divaricatus*) colonising salt muds of the Mediterranean region, susceptible to temporary inundation and extreme drying.
	Atlantic sea-pearlwort communities (*Saginion maritimae*): formations of annual pioneers occupying sands subject to variable salinity and humidity, on the coasts, in dune systems and salt marshes. They are usually limited to small areas and best developed in the zone of contact between dune and salt marsh.
	Central Eurasian crypsoid communities : Sparse solonchak formations of annual grasses of genus Crypsis (Heleochloa) colonizing drying muds of humid depressions of the salt steppes and saltmarshes (15.A) of Eurasia, from Pannonia to the Far East.
*Spartina swards (*Spartinion maritimae*, PAL.CLASS.: 15.2, 1320). Perennial pioneer grasslands of coastal salt muds are formed by Spartina or similar grasses. When selecting sites, preference should be given to those areas supporting rare or local* Spartina . *Sub-types are:*	
	Flat-leaved cordgrass swards: perennial pioneer grasslands of coastal salt muds, dominated by flat-leaved *Spartina maritima, S. townsendii, S. anglica, S. alterniflora*.
	Rush-leaved cordgrass swards: perennial pioneer grasslands of southern Iberian coastal salt muds, dominated by the junciform-leaved *Spartina densiflora*.
*Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*, PAL.CLASS.: 15.3, 1330)*	
Mediterranean and South Atlantic salt-marshes	

*Mediterranean salt meadows (*Juncetalia maritime*, PAL.CLASS.: 15.5, 1410)*	
	Tall rush (*Juncus maritimus* and/or *J. acutus*) dominated salt
	Short rush, sedge and clover saltmarshes (*Juncion maritimi*) and humid meadows behind the littoral, rich in annual plant species and in Fabacea (*Trifolion squamosi*)
	Mediterranean halo-psammophile meadows (*Plantaginion crassifoliae)*
	Iberian salt meadows (*Puccinellion fasciculatae*)
	Halophilous marshes along the coast and the coastal lagoons (*Puccinellion festuciformis*)
	Humid halophilous moors with the shrubby stratum dominated by Artemisia coerulescens
*Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*, PAL.CLASS.: 15.6, 1420). Perennial vegetation of marine saline muds mainly composed of scrub, essentially with a Mediterranean-Atlantic distribution*	
Halo-nitrophilous scrubs (Pegano-Salsoletea, PAL.CLASS.: 15.72, 1430), Halo-nitrophilous scrubs (matorrals) belonging to the Pegano-Salsoletea class, typical of dry soils under arid climates, sometimes including taller, denser bushes	
Mediterranean salt steppes (Limonietalia, PAL.CLASS.: 15.8, 1510)	
Boreal and Baltic salt-marshes	
	Agrostis stolonifera, Blysmus rufus, Bolboschoenus maritimus, Calamagrostis stricta, Carex nigra, C. paleacea, Centaurium littorale, C. pulchellum, Eleocharis uniglumis, E. parvula, Festuca rubra, Juncus gerardii, Odontites litoralis, Ophioglossum vulgatum, *Plantago maritima.*

II.2.2. EXAMPLES OF TEMPORAL VARIABILITY

Salt marshes can show large spatial and temporal variability. Due to the interaction of hydrodynamic forces and vegetation-sedimentation feedbacks, complex patterns of marsh establishment, development and destruction can occur at the same time spread across the marsh (see Ch. IV.2). Moreover, salt-marshes can go through cycles of large-scale marsh build-up alternated with lateral erosion resulting in a spatial shift of the leading vegetation edge and eroding cliff in time. Consequently, it will be a challenge to separate development trends in salt-marsh extant caused by environmental changes, by e.g. climate change and sea level rise, from changes due to the natural variability of this ecosystem.

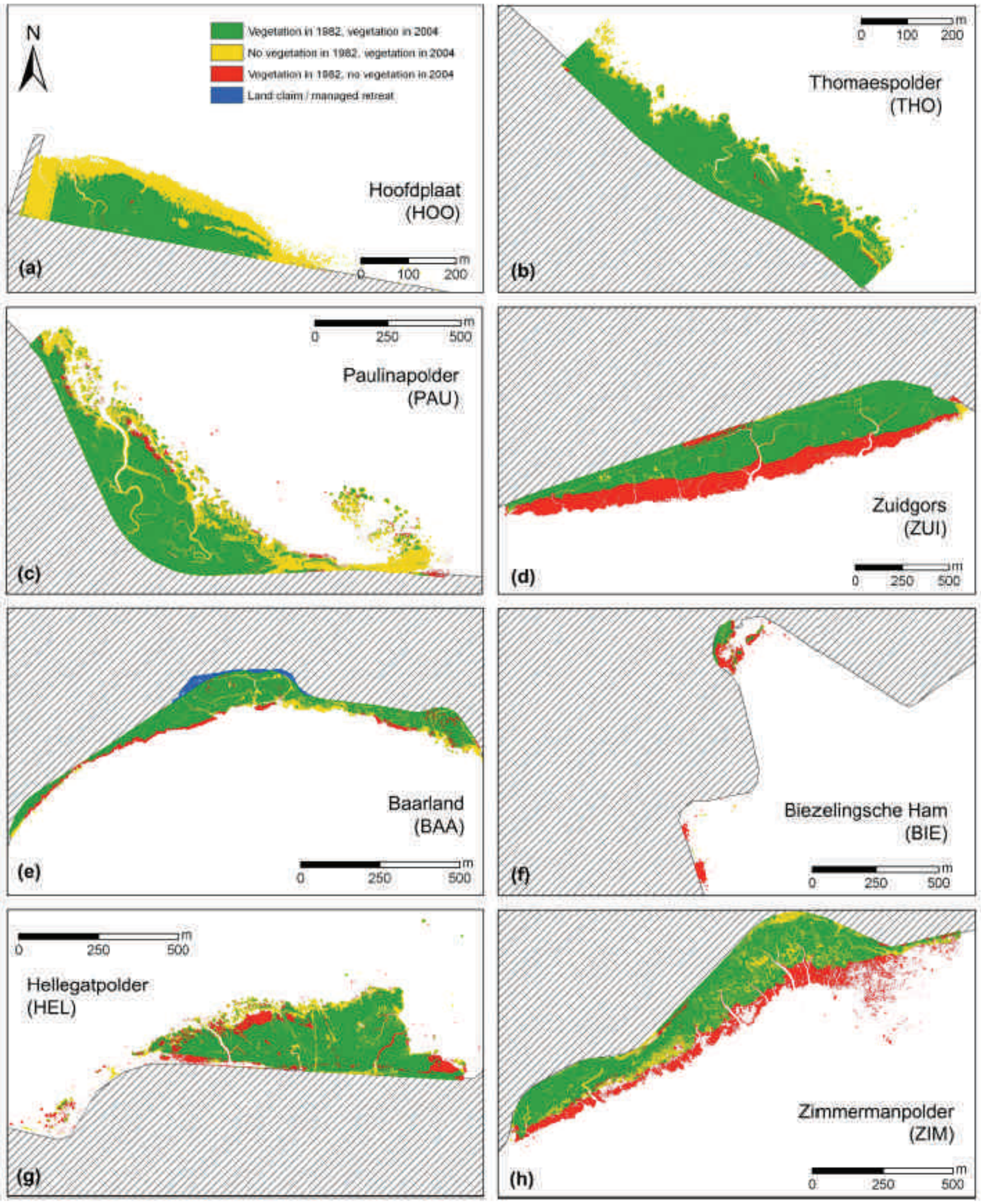


Figure 2.15. Spatiotemporal development of eight salt marshes along the Scheldt-estuary (figure taken from van der Wal et al., 2008).

Recently the spatiotemporal variability of eight salt marshes along the Scheldt estuary have been investigated (Fig.2.15, van der Wal., et al 2008). The study revealed the significance of intrinsic processes in salt marsh development, and the necessity to consider the local feedback mechanisms between plant growth, morphology and hydrodynamics of both the salt marsh and the mudflat, when assessing the status of salt marshes. Furthermore, the importance of assessing salt marsh changes in a spatial context is highlighted, rather than looking at changes in total salt marsh area. Most salt marshes showed simultaneous expansion of *Spartina anglica* tussocks, and lateral retreat of the mature salt marsh plateau, resulting in salt marsh rejuvenation, which support these conclusions. However, based on this study no clear relationship could be found between the net expansion or erosion and the hydrodynamic conditions at the edge of the salt marsh. Hence, additional mechanistic insight is required to determine which and how hydrodynamic and sedimentary conditions are important for the development and erosion of salt marshes (see ongoing research in V.2.1, where spatiotemporal variability of salt-marshes is correlated to fetch and sediment input by means of GIS and a modeling study).

The Yangtze Estuary is a typical medium tidal estuary with multi-order bifurcations, shoals and sand bars. Due to the distinct interactions between runoff and tidal currents, the hydrodynamic and geomorphologic processes of the estuary display a unique kinematical rule among the estuaries of the world. The enormous quantities of sediment produced by the Yangtze River have created extensive areas of shoals and tidal flats in the estuarine region, which have been colonized by various types of salt-marsh vegetation. Based on remote sensing mapping, the salt marsh vegetation in the Shanghai region amounted to 18314.8 ha (Year 2008).The tidal flats closest to the low water mark, elevation less than 2 m, are characterized by mud flats that are devoid of any vascular plants. As sedimentation and succession progressed, the *Phragmites australis* community replaced the *Scirpus mariqueter* community above the 2.9 m elevation. An additional species in this zone was *Spartina alterniflora*, which was introduced to the Yangtze Estuary in 1990's. Over the last two decades this species has gradually invaded large areas which were previously covered by *P. australis* and has also started to invade the upper parts of the *S. mariqueter* zone.

According to a recent study based on remote sensing mapping（Huang et al., 2008, the salt marsh vegetation in the Shanghai region amounted to 18314.8 ha (Year 2008) and more than 95% of saltmarsh vegetation belonged to the four major plant communities, i.e. the reed (*Phragmites australis*) community, the smooth cord-grass (*Spartina alterniflora*, an exotic species) community, and two types of sedge (*Scirpus mariqueter* and *Carex scabriflora*) communities. By their very nature, salt marshes and inter-tidal zones are dynamic and their biodiversity conservation and management relies on up to date spatial information, i.e. there is an urgent need for using remote sensing to map the salt marsh at a large spatial extent.

The study investigated the spatial and temporal dynamics of saltmarsh vegetation of Yangtze River Estuary wetlands between 1990 and 2008. A set of multi-temporal Landsat thematic mapper (TM) images were used, which covered the state of the low tide at the time the images were taken and the areas of different stages under the impacts of human activities. These satellite images were geometrically corrected by a series of nautical charts in the GIS platform using ESRI's Arc/GIS 8.1 software. Two spectral enhancement methods, Tasseled Cap (K-T) Transform and Normal Difference Vegetation Index (NDVI), were used to interpret satellite images more efficiently. Moreover, several *in situ* field surveys were carried out to revise the image classifications and for the classification accuracy assessment. The main results of this study were summarized in Table 2.7.

Year	1990	2000	2003	2008
Scirpus mariqueter	3781.5	4833.9	7602.2	4234.7
Phragmites. australis	14100.4	7085.6	10075.4	5617.5
Spartina alterniflora	0	1902.8	4275.8	5697.9
Others	0	0	0	2764.7
Total	17881.9	13822.2	21953.5	18314.8

Table 2.7. Changes of salt-marsh vegetation in Shanghai coastal wetlands during 1990-2008 (ha)

The total area of saltmarsh vegetation by 1990 in the Shanghai region was 17881.9 ha and *P. australis* community was the dominant species which accounted for 14100 ha and 80% of the total area of saltmarsh. However, almost half of *P. australis* saltmarsh had been lost during 1990 and 2000 as a result of the 1990s reclamations. At the same time, *S. alterniflora* began to establish and invade to the wetlands. The total area of saltmarsh vegetation had reached to 21953 ha and the area of *S. alterniflora* to 4276 ha by 2003. After 2003, the total area of saltmarsh decreased mainly due to intense reclamation. Both *S. mariqueter* and *P. australis* communities declined almost by half, while *S. alterniflora* community was continuing to expand and reached 5698 ha and accounted for more than one quarter of the total area of saltmarsh vegetation by 2008.

In the Chongming Dontan national nature reserve (the total area covers 24200 ha, including 10000 ha and 14000 ha of tidal flats above and below the elevation of 0 m, respectively), *P. australis* community experienced degradation since 1990 mainly because of the reclamations in 1992, 1998 and 2001. Subsequently, the *P. australis* community began to increase slowly. From 1990 the area of *P. australis* community had kept increasing for more than 15 years. With the rapid range expansion of *S. alterniflora*, the area of *P. australis* began to decrease since 2005, but the area of this species was always maintained at greater than 2000 ha. The exotic species *S. alterniflora* continued expanding since its initial introduction, and had accounted for almost one third of the total intertidal saltmarsh at the Chongming Dongtan nature reserve by 2008.

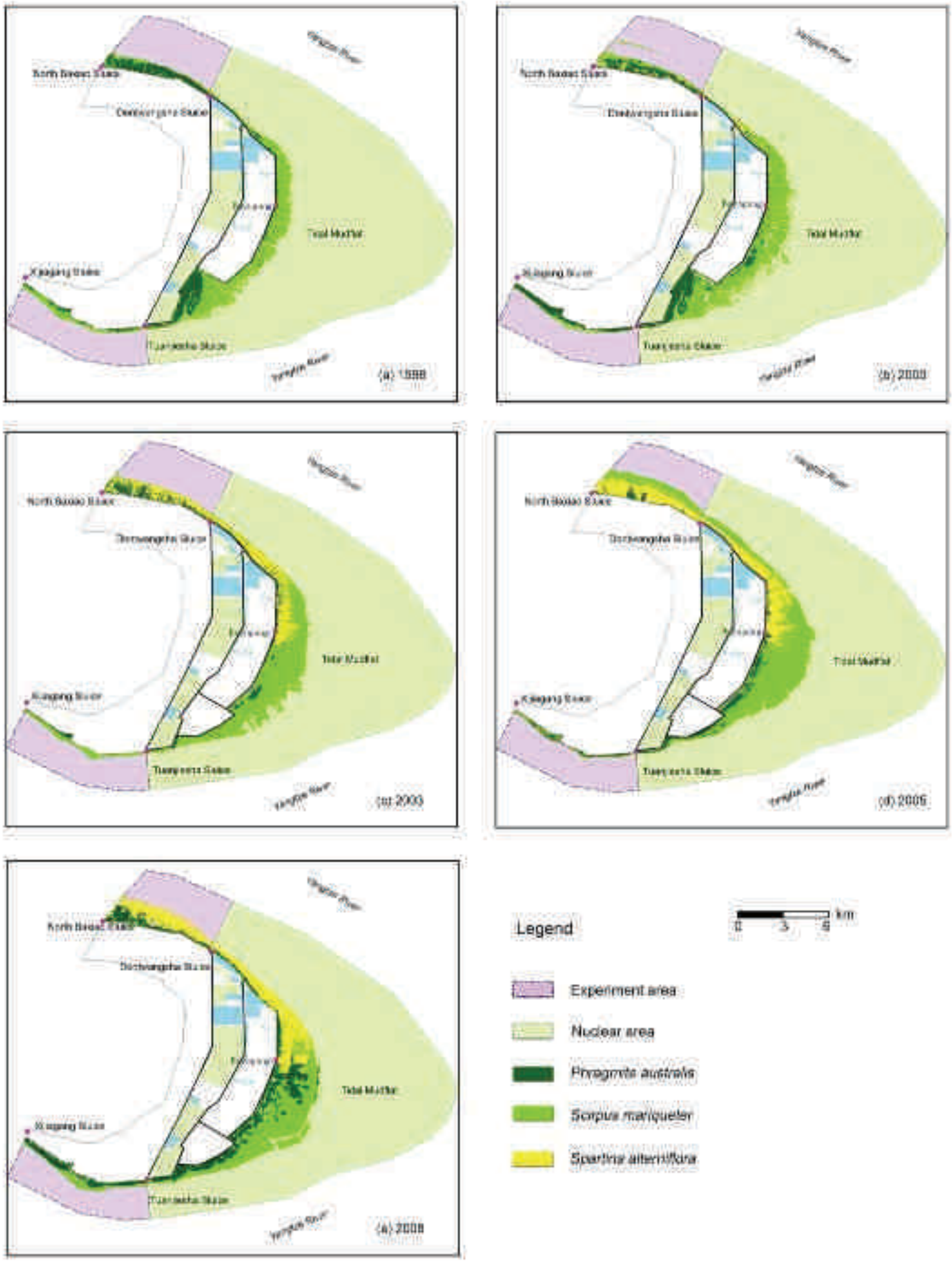


Figure 2.16.The spatio-temporal dynamics of salt marshes vegetation and invasion history of *Spartina alterniflora* during in 1998-2008 at the Chongming Dongtan nature reserve.

The Jiuduansha national nature reserve (the total area covers 423 km^2). Jiuduansha Shoals are new formed neonatal islands and grow very fast due to their unique location at Yangtze Estuary. The

Jiuduansha shoals have never been colonized by humans and have been in a natural condition since they were formed. Since 1987, the pioneer *Scirpus mariqueter* saltmarsh community could be discerned from satellite images, which showed the community first colonizing on the tidal flats above the 2 m elevation. As an ecological engineering project carried out by the State Key Laboratory of Estuarine and Coastal Research, 40 ha of *Phragmites australis* and 55 ha of *Spartina alterniflora* were planted on the Jiuduansha shoals in 1997. From 1997 to 2008, the total salt marsh area had increased from 1094.6 ha to 3600.6 ha, with a mean increasing rate of 230 ha/a. *S. mariqueter* increased from 966.6 ha in 1997 to 1850.2 ha in 2003, whereas the area of *S. mariqueter* began to decrease afterwards with the rapid range expansion of *S. alterniflora* population in Middle Shoal and Lower Shoal. The mean increasing speed for *P. australis* community was 70 ha/a, and the area increased from 168 ha in 1997 to 924 ha in 2008. *S. alterniflora* had showed strong competitive capacity at Jiuduansha Shoals. Its expanding rate exceeded any of the indigenous species. The area of *S. alterniflora* had increased to 1708.6 ha by 2008, accounting for 47.5% of the total salt marsh vegetation, and had dominated the largest area on Jiuduansha Shoals.

TM date	*S. alterniflora*	*P. australis*	*S. mariqueter*	Total
1997-10-20	55*	167.5	966.56	1094.06
2000-5-24	101.61	353.79	1017.09	1472.49
2001-7-26	283.71	368.91	1382.85	2035.47
2002-11-11	377.06	401.94	1608.22	2387.215
2003-8-2	469.62	463.41	1850.22	2783.25
2004-7-19	1014.39	563.49	1789.02	3366.9
2005-11-27	1281.01	637.89	1493.70	3412.6
2008-04-25	1708.57	924.00	968.03	3600.60

*Introduced in 1997

Table 2.8. The spatial-temporal dynamics of saltmarsh vegetation on the intertidal zones of Jiuduansha Shoals (ha)

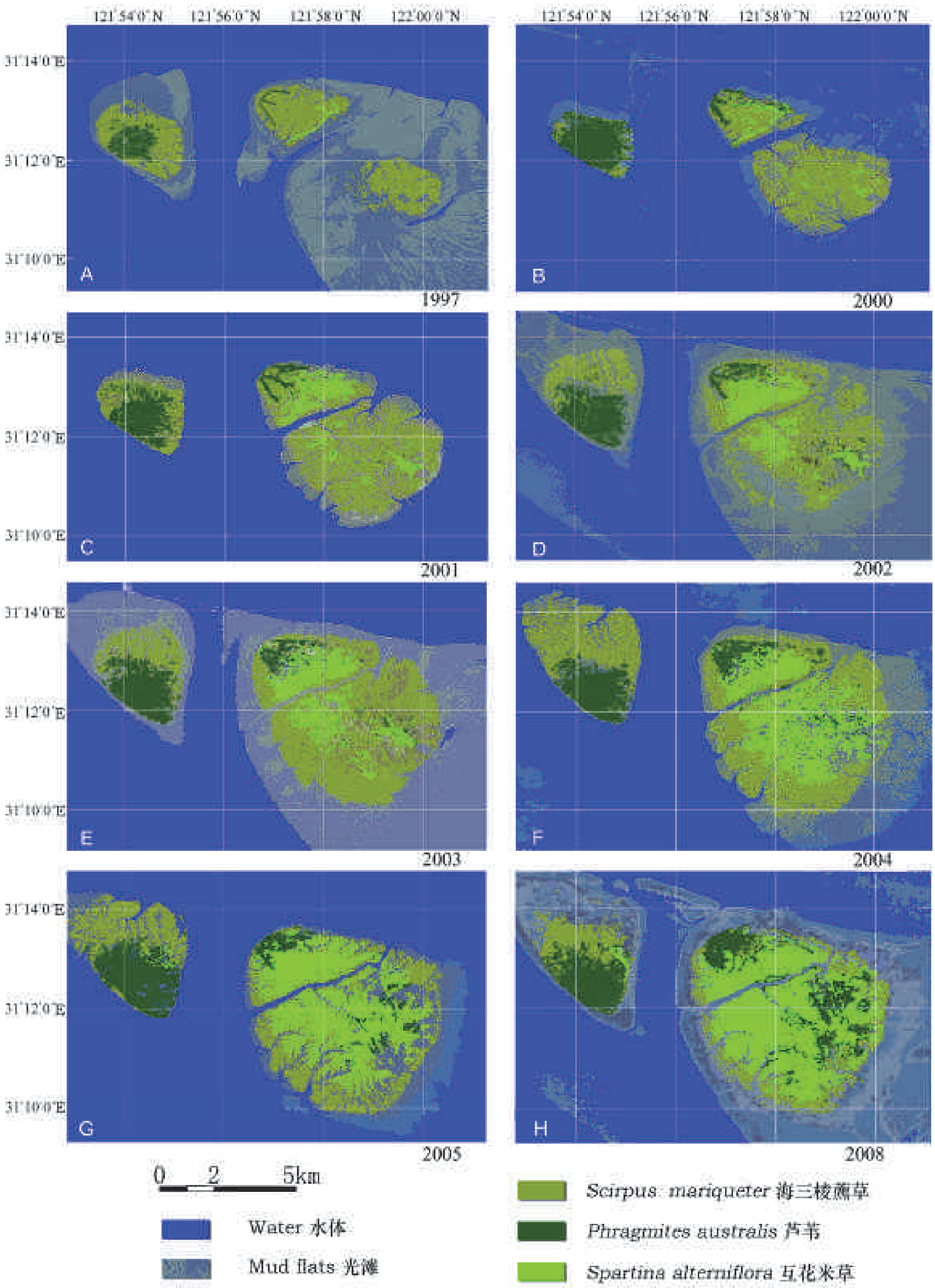


Figure 2.17. The Spatio-temporal dynamics of salt marsh vegetation at Jiuduansha Shoals

Expansion of *S. alterniflora*, an invasive species, on the intertidal mudflats of Jiuduansha Shoals at the Yangtze Estuary is a prime example of a spatially-structured invasion in a relatively simple habitat, for which strategic control efforts can be modeled and applied. Here, we developed a Cellular Automata (CA) model, in conjunction with Remote Sensing and Geographical Information Systems, to simulate the expanding process of *S. alterniflora* for a period of 10 years after being introduced to the new shoals, and to study the interactions between spatial pattern and ecosystem processes for the saltmarsh vegetation. The population expansion pattern of *S. alterniflora* was compatible with the common feature of invasion, i.e. the initial colonization, a lag time and the rapid range expansion. The results showed that the CA model could simulate the population dynamics of *S. alterniflora* and *P. australis* on the Jiuduansha Shoals successfully. The results strongly support the hypothesis of space pre-emption as well as range expansion with simple advancing wave fronts for these two species. In the Yangtze River Estuary, the native species *P. australis* shares the same niche with the exotic species *S. alterniflora*. However, the range expansion rate of *P. australis* was much slower than that of *S. alterniflora*. With the accretion of the Jiuduansha Shoals due to the large quantity of sediments deposited by the Yangtze River, a rapid range expansion of *S. alterniflora* is predicted to last for a long period into future.

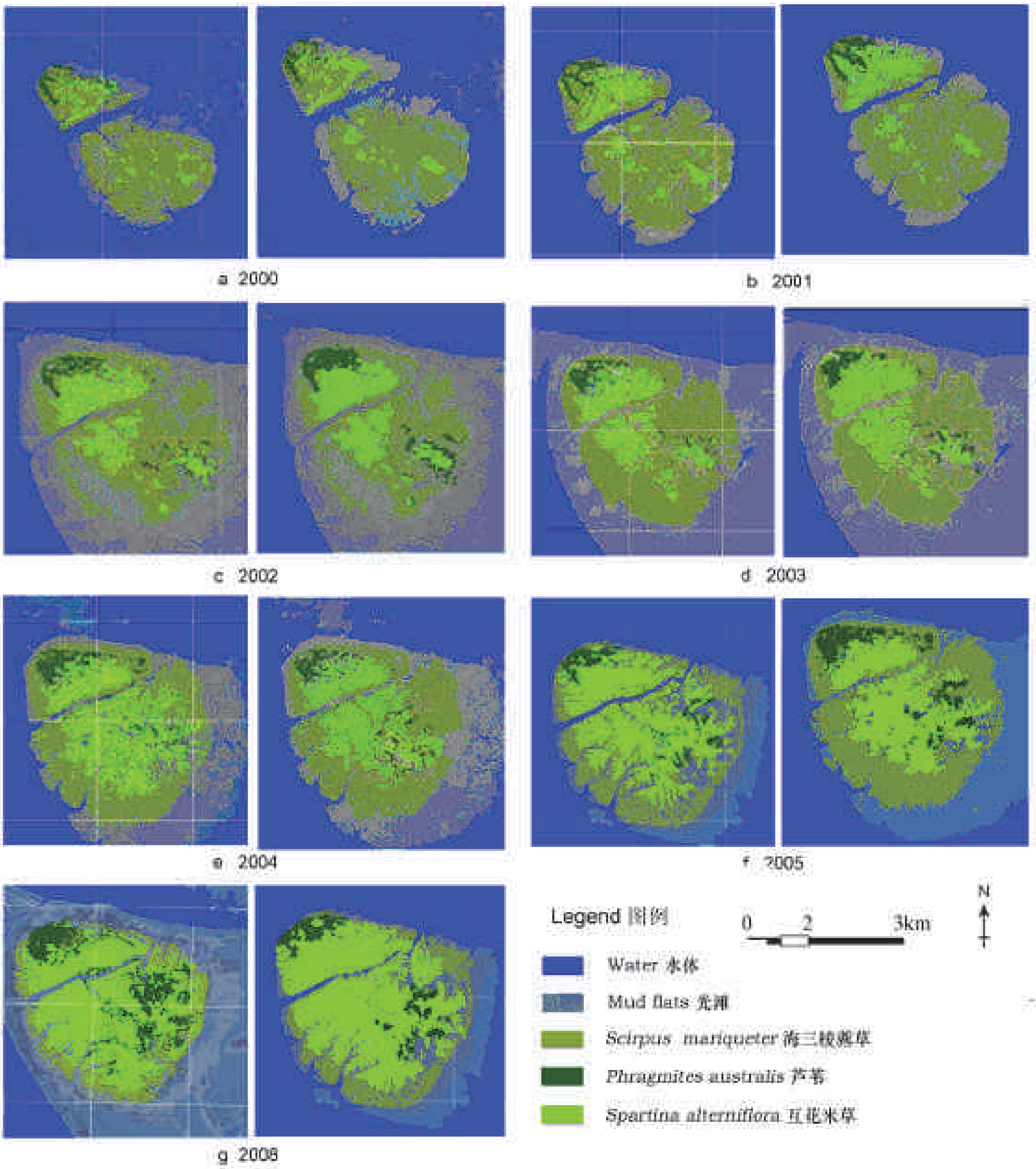


Figure 2.18. The results of CA simulation of *Phragmites australis* and *Spartina alterniflora* population dynamics on Jiuduansha Shoals (a. the image classification (left) and simulation (right).

II.3. BIOGENIC REEFS

[Based on text written for ID3.1]

II.3.1. EUROPEAN-SCALE DISTRIBUTION OF BIOGENIC REEFS

Biogenic reefs can be described as are hard compact structures created by the activity of living organisms. They do not share a uniform structure and are found at variable spatial scales. Dense colonies of several species are widely considered to be reef in Europe. Only four of these species are described in this report due to their contribution to sediment entrainment, bed stability and potential wave energy attenuation, these are: *Sabellaria alveolata*, *Sabellaria spinulosa*, *Modiolus modiolus* and *Mytilus edulis*. Many biogenic reef habitats are currently threatened and/or are in decline in Europe as a result of various natural and anthropogenic pressures (OSPAR 2010). Figure 2.19 Illustrates the distribution of some biogenic reef habitats which are currently in decline around the coast of Europe.

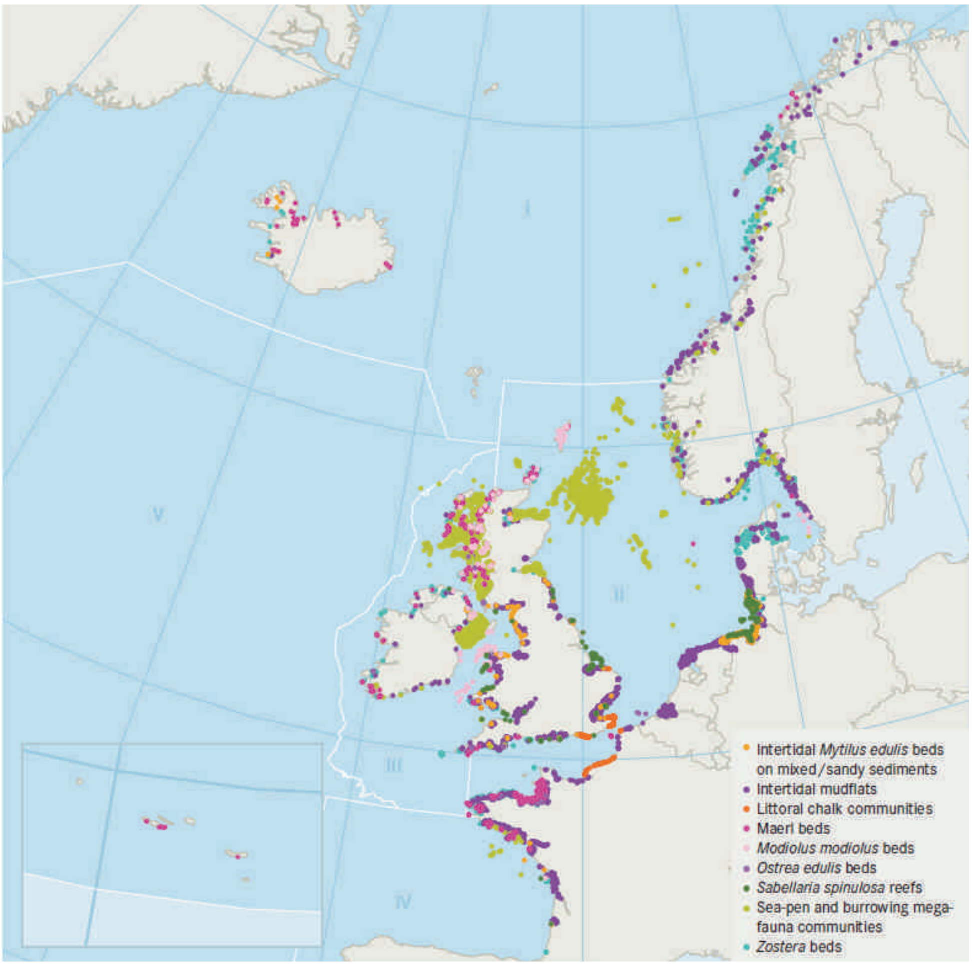


Figure 2.19. Map taken from the OSPAR Quality Status Report 2010 depicting the distribution of threatened and/or declining coastal and shelf habitats in Europe.

Sabellaria spinulosa

EUNIS code A4.22 and A5.611

Sabellaria spinulosa reefs are known from all European coasts, except the Baltic and the waters of the Kattegat and Skagerrak, but are typically limited to areas with very high levels of suspended sediment (OSPAR 2010, Fig. 2.20). In the UK aggregations of *S. spinulosa* are reported to occur at a number of locations around the British Isles (Holt et al., 1998; Davies et al., 2009). Perhaps the best known example of an *S. spinulosa* reef in the UK is found in the mouth of the Wash (east coast of England), where reefs are elevated above the seafloor and have been found to extend over hundreds of square meters within the Norfolk Coast SAC (Foster-Smith & Hendrick 2003). Relatively few records have been found in Scotland (Fig. 2.19). Not all of these aggregations could be described as "reefs", for instance where the species may only form superficial crusts on mixed substrata. On the German coast, intertidal and subtidal reefs have been reported from the Wadden Sea (Berghahn & Vorberg, 1993) and from the southern North Sea where Linke (1951) reported reefs up to 60 cm thick, 8m wide and 60 m long. *S. spinulosa* has also been reported from the French coast, but without precise locations (Holt et al., 1998).

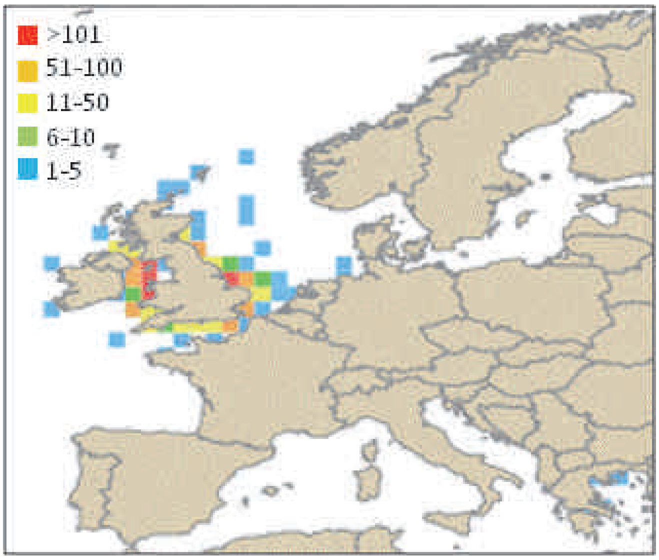


Figure 2.20: Current OBIS distribution data for *S. spinulosa* in Europe (data from OBIS, April 2011). Please note that older records and those from southern Europe are probably missing.

Sabellaria alveolata

EUNIS code A2.711

The records of *Sabellaria alveolata* throughout Europe are greater in northern latitudes (Fig.2.21). This is an obvious artefact of data reporting to OBIS as *S. alveolata* has been reported to be widely distributed in the France, Spain and Portugal and extends as far south as Morocco (Gruet 1982; Cunningham et al., 1984). It reaches its northern limits in Britain but is restricted to the warmer waters off the west coast as growth is inhibited below 5°C (Crisp 1964). The current confirmed northern limit is the Dumfriesshire coast of SW Scotland with records needing confirmation from the Firth of Clyde and Outer Hebrides. This species builds the largest reefs on the European coast; in particular the “Les Hermelles” reef in the Saint-Michael Bay in France is over 100 ha and is considered the largest reef in Europe (Gruet 1982; Marchand & Cazoulat 2003).

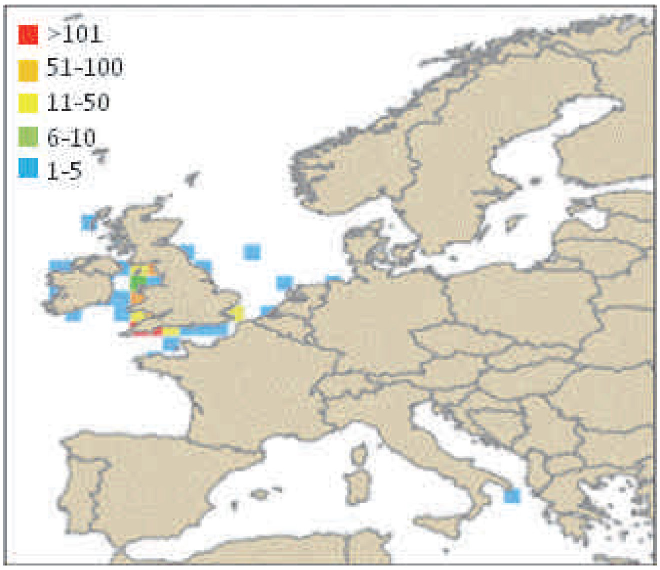


Figure 2.21: Current OBIS distribution data for *S. alveolata* in Europe (data from OBIS, April 2011) showing distributions and unconfirmed records. Please note that older records and those from southern Europe are probably missing.

Intertidal *Mytilus edulis*

EUNIS Code: A2.7211 and A2.7212

The distribution of *Mytilus edulis* is circumpolar in boreal and temperate waters, in both the southern and northern hemispheres extending from the Arctic to the Mediterranean in the north-east Atlantic (Soot-Ryen 1955). The majority of intertidal beds are found in the Wadden Sea (Netherlands, Germany and Denmark) where a 2007 inventory reported an estimated coverage of 1865 hectares in the Dutch sector (Goudswaard et al., 2007). It is also present in British coastal waters, Ireland (Jones et al., 2000) and there is a large bed (covering approximately 200 ha) in southern Brittany in France (Rollet et al., 2005).

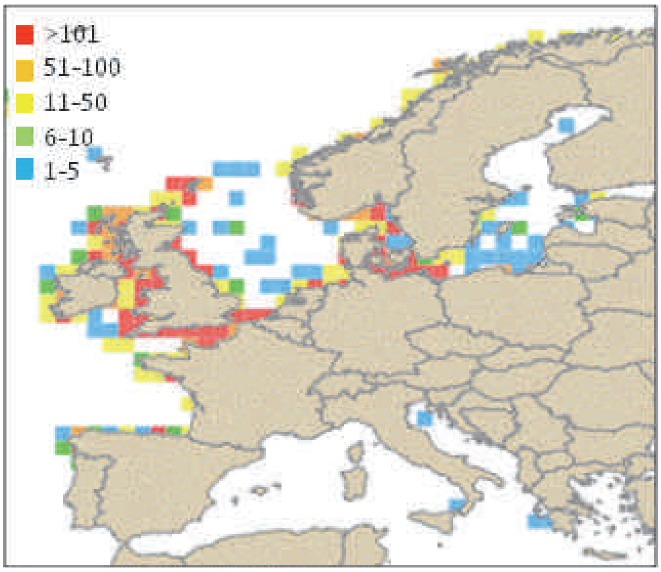


Figure 2.226: Current OBIS distribution data for *Mytilus edulis* in Europe (data from OBIS, April 2011). Please note that older records and those from southern Europe are probably missing and that *M. galloprovincialis* is widespread throughout the Mediterranean.

Modiolus modiolus

EUNIS codes A5.621, A5.622, A5.623 and A5.624

Modiolus modiolus is an Arctic-boreal species that is limited in distribution by warmer temperatures to the south but occasionally specimens have been reported as far south as north west Africa. It occurs from the Bay of Biscay to northern Norway, with occurrences off Iceland and the Faeroes (Tebble 1966; Poppe & Gotö 1993). It is found throughout British waters, but has most frequently been reported in northern and western areas (Fig. 2.23). Extensive horse mussel beds are found only in parts of north and western Scotland, the Ards Peninsula, Strangford Lough, the Isle of Man, north-west Anglesey and north of the Lleyn Peninsula.

Descriptions of M. modiolus usually state the presence of aggregated clumps on mud or muddy-gravel sediments although the vast majority of these will not fall into the definition of biogenic reef, due to low density and coverage. However, several areas do contain large beds definable as biogenic reef including beds in Strangford Lough (Roberts 1975), the Isle of Man (Jones 1951; unpublished references in Holt et al., 1998), Scottish waters (Comely 1978; Howson et al., 1994) and within the Lleyn Peninsula (Lindenbaum et al., 2008). One notable area of horse mussel beds that has received significant research focus have been those within the Bay of Fundy on the Scotian Shelf, Canada (see Wildish et al., 2009).

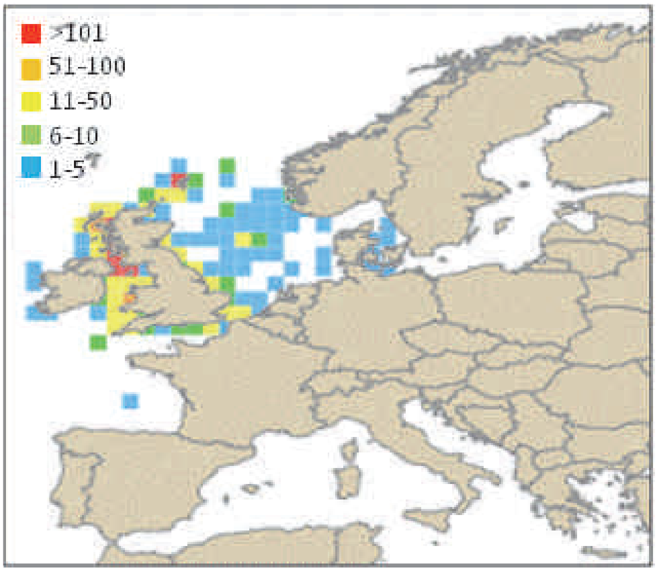


Figure 2.23. Current OBIS distribution data for *Modiolus modiolus* in Europe (data from OBIS, April 2011). Please note that older records and those from southern Europe are probably missing.

II.3.2. EXAMPLES OF TEMPORAL VARIABILITY

Sabellaria alveolata

Cunningham et al. (1984) reviewed the distribution and local abundance of *S. alveolata* in Britain. This review used past records from the literature, data from new shore surveys and reports via correspondence from other marine scientists. As a result of this exercise, changes in the extent of *S. alveolata* distribution over a period of approximately 100 years were documented. In order to evaluate long-term temporal variability in *S. alveolata* distribution and abundance, the data were divided into three arbitrary periods: pre-1963 (before the cold winter of 1962/1963), 1964-1979 and 1980-1984 (Cunningham et al., 1984).

Frost et al. (2005) carried out a series of broadscale and focused mapping studies of *S. alveolata* in NW England and North Wales in 2003/04. This comprised a resurvey of sites that had been previously surveyed in the 1980s (Cunningham et al. 1984). *S. alveolata* was found to be present at most of the sites where it had previously been recorded (e.g. Cunningham, 1984) and at many of these sites it appears also to have increased in abundance (Table 2.9). *S. alveolata* had re-appeared in areas where it has been absent for many years (Table 2.9: Hilbre Island and Colwyn Bay) and had spread to areas for which there are no known previous records (Table 2.9: North Wirral, Rossal Point).

Hawkins (1993) suggested that *S. alveolata* was declining along the Cumbrian coast but the present study found it to be abundant or super-abundant at most sites. The records from the present study therefore seem to confirm the observation made by others that *S. alveolata* shows a great deal of temporal variability within a fairly constant geographic range (e.g. Cunningham et al., 1984). Even on a shore where *S. alveolata* is continually present, there is a great deal of variability in terms of abundance and 'within shore' distribution. For example, long term studies at Duckpool in North Cornwall (Wilson

1971; 1974; 1976) and in Normandy, France (Gruet 1986) have revealed a great deal of variability over the years in the distribution and abundance of *S. alveolata* colonies within sites.

Location	**OS Reference**	**S. alveolata abundance**			
		Pre-1963	**1964-1979**	**1980-1984**	**2003-2004**
Penmon	SH634805			N	N
Great Orme's Head	SH7583			N	N
Little Orme's Head	SH820825			N	N
Rhos-on-Sea	SH842812			N	N
Colwyn Bay	SH8778	P		N	R
Hilbre Island	SJ1887	A	R	N	A
Wirral Foreshore					A
Lytham Pier	SD3327			N	N
St Annes Pier	SD3129			N	N
Fleetwood, Rossall Pt	SD312479			N	F
Heysham*	SD4060	F-O		N	N
Holme Island	SD4060			N	N
Humphrey Head	SD421780			N	N
Wadhead, Scar	SD392732			N	N
Walney Island	SD313744			N	N
Annaside Bank	SD090849			A	SA
Tarn Bay	SD073903			A-SA	SA
Drigg	SD041983			A	SA
Seascale	NY032011			O	SA
Sellafield	NY0103			O	A-SA
Nethertown	NX986073			A	A
St. Bees	NX955117			O	C-A

Table 2.9. Past data on *Sabellaria alveolata* maximum abundance in Northwest England and Wales, with recent resurveys included. N = absent, R = rare, O = occasional, F = frequent, C = common, A = abundant and SA = super-abundant (massive reefs). P = recorded as present but abundance not known. From Cunningham et al. (1984) and Frost et al. (2005).

Modiolus modiolus

Only a few beds are known have been surveyed over long enough time spans for evidence of change to be apparent. In the Irish Sea, south of the Isle of Man, an extensive bed was almost completely lost due to scallop dredging (Veale et al., 2000). For similar reasons, beds in Strangford Lough (Northern Ireland) also showed severe declines (Service & Magorrian 1997). Recently beds in North Anglesey (Wales) have been destroyed by fishing activity (Holt 2008, Countryside Council for Wales, pers. comm.). By contrast, in an Icelandic bay *Modiolus modiolus* was still the dominant by-catch species in scallop dredges 30 years after scallop dredging began (Garcia & Ragnarsson 2007). In Sullom Voe (Shetland) a bed coincident with a pipeline showed signs of recovery, with some re-colonisation of disturbed sediment after a few years (Mair et al. 2000). On the legs of an oil platform in the North Sea a substantial population was present 10 years after installation, but in this situation the young mussels would have been free of much predation (Anwar et al. 1990). As a species it appears to have declined in the North Sea. Comparing occurrences by ICES Rectangles Callaway et al. (2007) showed that the species had been found in the 1982-85 period in 11 rectangles, but comparable international surveys in 2000 found it in only 1 rectangle.

Sabellaria spinulosa

Subtidal *S. spinulosa* reefs have been reported to have been lost in at least five areas of the north east Atlantic (Jones et al., 2000). During the 1920s large reefs of *S. spinulosa* were common in the German Wadden Sea (Hagmeier & Kändler 1927) but most have since been lost. Similar records of loss have been recorded from the Lister Ley (Island of Sylt) and the Norderau area (Riesen & Reise 1982; Reise & Schubert 1987). Only three living reefs were found during surveys in the early 1990s compared to 24 during the 19th century (Fig. 2.24). In the late 1990s, samples taken from the subtidal reefs in the German Wadden Sea consisted largely of compact lumps of empty tubes. In 2000, one of these reefs had diminished drastically in extent with the remainder in poor condition although dredge samples were occupied by many tiny tubes with living worms inside. A third reef which had previously extended over ~18 hectares could not be located during repeat surveys in 2002. In the UK there are reports of reefs being lost in Morecambe Bay (Taylor & Parker 1993), the Wash and the Thames (Warren & Sheldon 1967). In the western North Sea a report comparing records from 1986 and 2000 suggest an increase in distribution and densities in the western North Sea (Rees 2007).

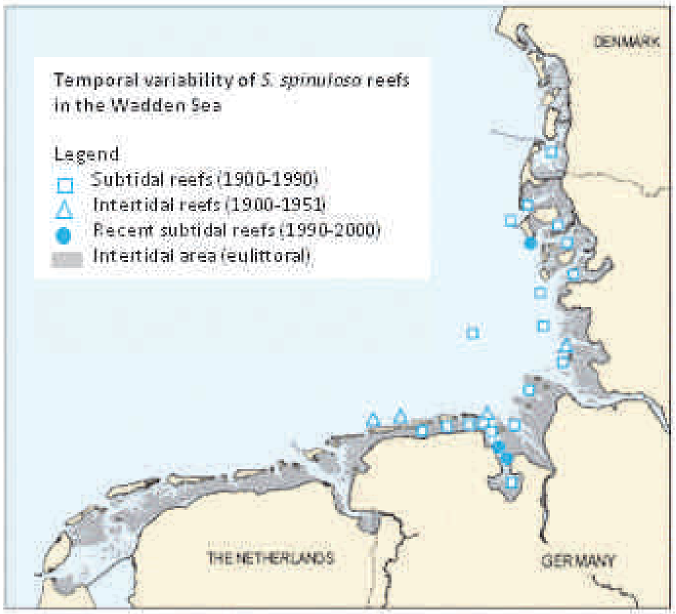


Figure 2.24. Changing occurrences of *S. spinulosa* reefs in the Wadden Sea (Wadden Sea Secretariat, 2005). Figure adapted from OSPAR 2010.

Mytilus edulis

Surveys covering the whole littoral of Niedersachsen, in Germany, revealed a decrease in the extent of *M. edulis* (5000 hectares in the late 1950s, 2700 ha in 1989/91, 1300 ha in 1994 to 170 ha in 1996). Mussel beds in the Ameland region have also disappeared after intensive fishing in the region (Dankers 1993). In the Netherlands, Higler *et al*. (1998) observed a serious decline in the populations of mussels between 1988 and 1990, mainly caused by fisheries. The extent of mussel beds decreased from the 1970s to the 1990s. In Denmark, intensive fisheries during 1984 to 1987 almost led to a complete disappearance of the mussel population (Kristensen 1995).

III. COASTAL PROTECTION VALUE OF DUNES, SALT MARSHES & BIOGENIC REEFS: A LITERATURE REVIEW AND OVERVIEW OF ONGOING AND FINISHED MEASUREMENTS[2]

III.1. DUNES

III.1.1. LITERATURE REVIEW

[Based on text written for OD2.1]

Dunes play an important role in defence against storms and protect infrastructures and ecosystems particularly during severe winter storms. During storms, coastal dunes erode and nourish front and down drift beaches and near-shore sand bars. Sand bars, beaches and dunes interact, exchanging sand. This interaction dissipates storm waves energy.

Dunes act like a sand stock for eroded beaches (Fig. 3.1).

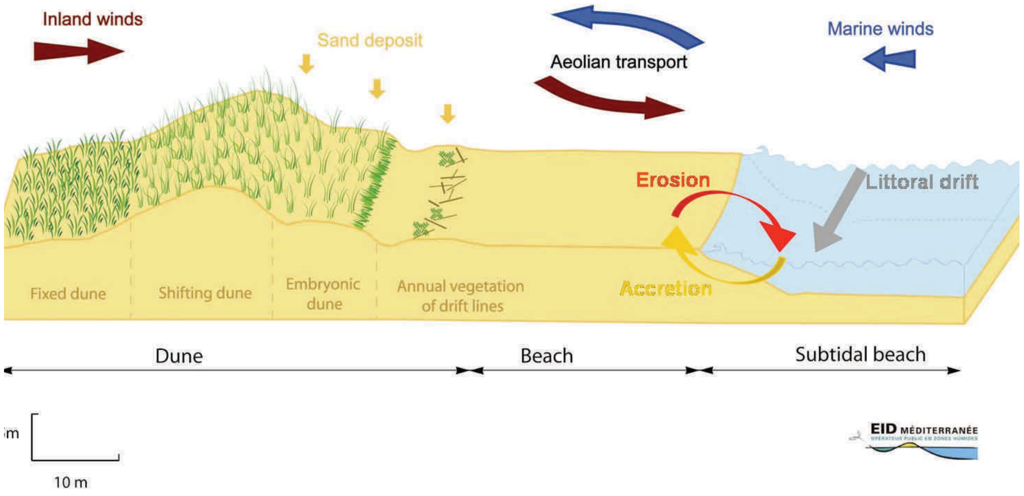


Figure 3.1.: Dune section

Coastal flooding linked to sea surges are likely to increase during the 21th century (Sabatier et al. 2007; Ullmann et al., 2007)in response to increasing storminess (Ullmann and Pirazzoli, 2007) and rising sea levels (Suanez et al., 1997). These temporary submersions (a couple of hours to a couple of days) are limited to the beach when dunes are wide enough to ensure protection for the back areas. Dunes act as a barrier to storm surges and flooding, protecting landward development and natural ecosystems

(Paskoff, 1998). Dunes absorb the impact of high-energy storms preventing or delaying intrusions of water into inland areas (Gómez-Pina et al., 2002).

The height of the dune must be close to the adjacent dunes, and of sufficient height to limit the risk of marine submersion. It is necessary to carry out gentle slopes: the profile of the dune must be the least aggressive possible with the wind action (Heurtefeux et al., 2007). Lastly, shaping must take account of the local dynamic context.

But dune formation is linked to natural obstacles. It is well documented (Halpern et al., 2007) that dune vegetation is specifically adapted to take advantage of wind sediment transport. It has been observed that coastal plants capture sediments, elevate the substrate which allows the vegetation to grow strong and vigorous (Tsoar, 2005), as we can see in the following picture.

Figure 3.2. Sand accumulation at the bottom of *Ammophila arenaria*

There is a real interest in the ecological engineering capabilities of coastal plants, yet only a few studies have explored the eco-geomorphological limits of these capabilities, that is why further analysis will be done within the THESEUS experimentations.

Without dune plants, the integrity and preservation of a stable dune complex cannot exist. Dune plants are especially sensitive to disturbance and are heavily affected by humans. Anthropogenic processes combined with the natural regression of the coastal line, induce the acceleration of the destruction of the dune vegetation (Araujo et al., 2002). Changes in vegetation structures influence habitat sustainability and associated faunal community structure. In addition to their economical relevance, coastal dunes have great intrinsic value that must be protected from further losses and fragmentation due to human influence. Experimentation throughout the THESEUS project will provide information for developing a tool for assessing the conservation status of coastal sand dune ecosystems in order to assess their protection value.

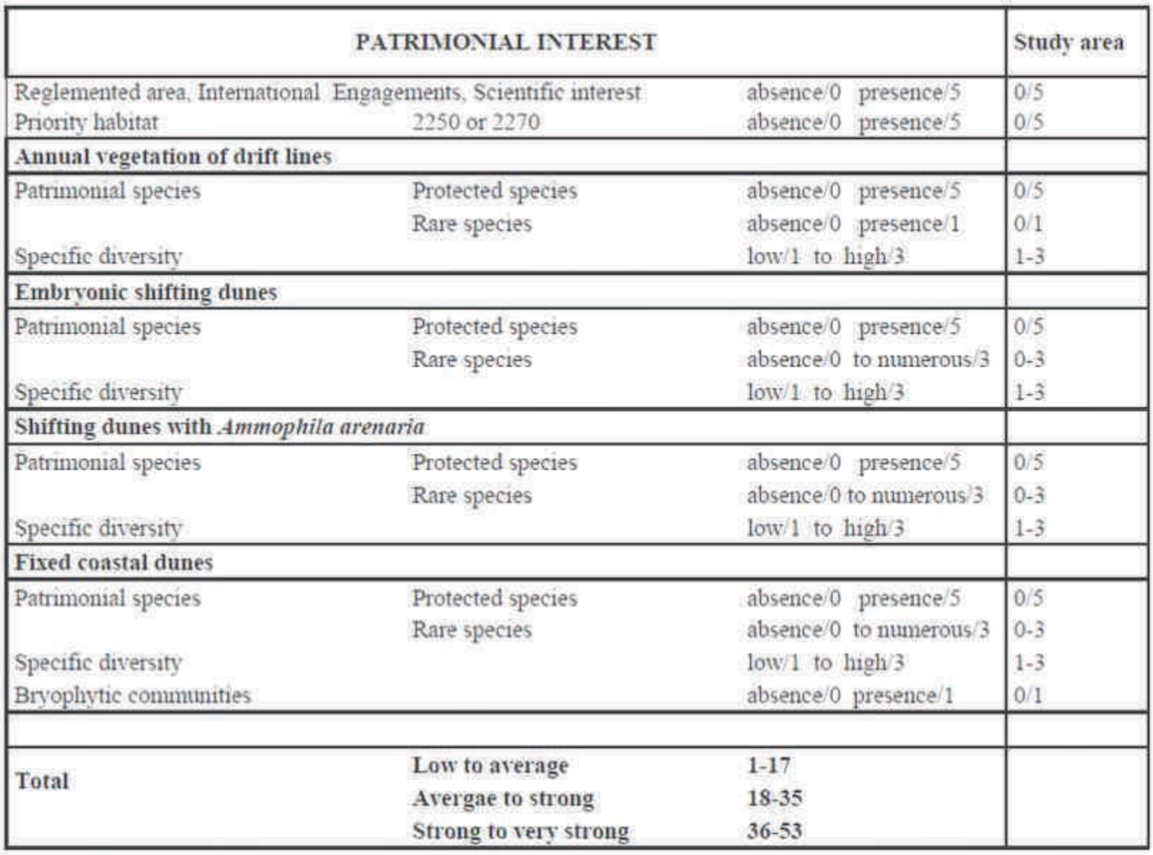
III.1.2. MEASUREMENTS WITHIN THESEUS

We develop a global assessment method in order to standardise the evaluation of dune patrimonial interest, and conservation status. Sandy seashores show an extremely specialized flora which has developed many adaptations to the arid conditions on dunes. Coastal dune systems represent a mosaic of natural habitats and so a big biodiversity. Without vegetation, the integrity and preservation of stable dune complex cannot exist. We attempt to identify best descriptors based on vegetation for this tool in Establishment of criteria for a comprehensive assessment method of Languedoc-Roussillon coastal dunes (Grosset, EID Méditerrannée, 2007). The aim is to develop a field tool that provides information synthesis and help for global monitoring and management of coastal dune. This tool was developed and tested only on the Languedoc-Roussillon coastal dunes but it could be easily adapted to any dune slack.

On dunes, plants are submitted to hard conditions: high temperatures, strong winds, sand input, ground instability, salinity and low pluviometry. According to a degree of exposure to those conditions, it appears different zones for vegetal communities.

To carry out this assessment, it is important to begin by covering the entire dune system and to choose a representative area. The evaluation must always occur at the same period, during spring. Rare taxa are searched on the setting, thanks to a list of patrimonial species that can be found on coastal dunes. Regardless of their number, the presence of one protected taxa mark out 5. Concerning rare taxa, the evaluation is based on the number of species and their rarity level. Specific diversity must be evaluated only with the native and characteristic dunes plants.

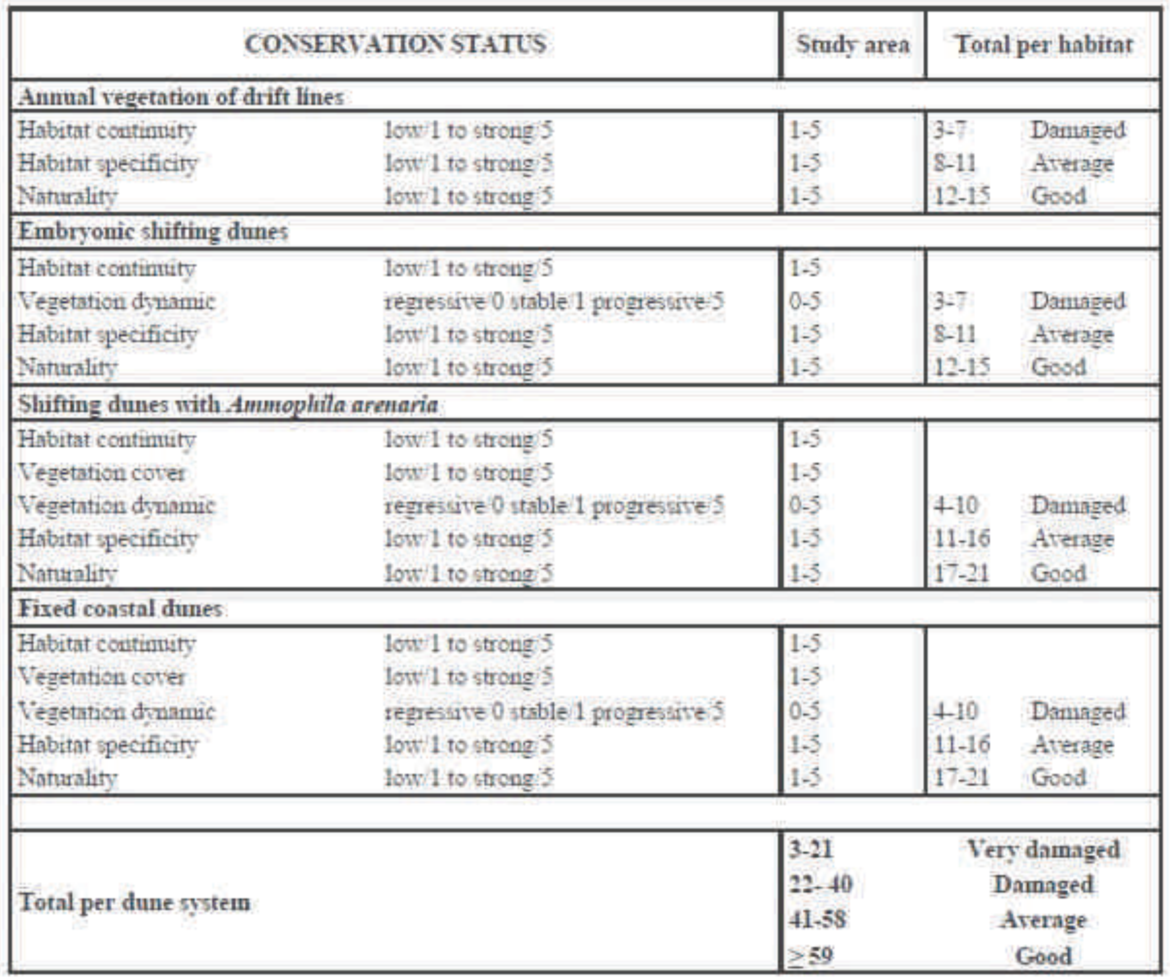
Patrimonial value assessment



PATRIMONIAL INTEREST			Study area
Reglemented area, International Engagements, Scientific interest		absence/0 presence/5	0/5
Priority habitat	2250 or 2270	absence/0 presence/5	0/5
Annual vegetation of drift lines			
Patrimonial species	Protected species	absence/0 presence/5	0/5
	Rare species	absence/0 presence/1	0/1
Specific diversity		low/1 to high/3	1-3
Embryonic shifting dunes			
Patrimonial species	Protected species	absence/0 presence/5	0/5
	Rare species	absence/0 to numerous/3	0-3
Specific diversity		low/1 to high/3	1-3
Shifting dunes with *Ammophila arenaria*			
Patrimonial species	Protected species	absence/0 presence/5	0/5
	Rare species	absence/0 to numerous/3	0-3
Specific diversity		low/1 to high/3	1-3
Fixed coastal dunes			
Patrimonial species	Protected species	absence/0 presence/5	0/5
	Rare species	absence/0 to numerous/3	0-3
Specific diversity		low/1 to high/3	1-3
Bryophytic communities		absence/0 presence/1	0/1
Total	**Low to average**	**1-17**	
	Avergae to strong	**18-35**	
	Strong to very strong	**36-53**	

Table 3.1.Patrimonial interest evaluation grid

Conservation status



CONSERVATION STATUS		Study area	Total per habitat	
Annual vegetation of drift lines				
Habitat continuity	low/1 to strong/5	1-5	3-7	Damaged
Habitat specificity	low/1 to strong/5	1-5	8-11	Average
Naturality	low/1 to strong/5	1-5	12-15	Good
Embryonic shifting dunes				
Habitat continuity	low/1 to strong/5	1-5		
Vegetation dynamic	regressive/0 stable/1 progressive/5	0-5	3-7	Damaged
Habitat specificity	low/1 to strong/5	1-5	8-11	Average
Naturality	low/1 to strong/5	1-5	12-15	Good
Shifting dunes with *Ammophila arenaria*				
Habitat continuity	low/1 to strong/5	1-5		
Vegetation cover	low/1 to strong/5	1-5		
Vegetation dynamic	regressive/0 stable/1 progressive/5	0-5	4-10	Damaged
Habitat specificity	low/1 to strong/5	1-5	11-16	Average
Naturality	low/1 to strong/5	1-5	17-21	Good
Fixed coastal dunes				
Habitat continuity	low/1 to strong/5	1-5		
Vegetation cover	low/1 to strong/5	1-5		
Vegetation dynamic	regressive/0 stable/1 progressive/5	0-5	4-10	Damaged
Habitat specificity	low/1 to strong/5	1-5	11-16	Average
Naturality	low/1 to strong/5	1-5	17-21	Good
Total per dune system		**3-21**	**Very damaged**	
		22-40	**Damaged**	
		41-58	**Average**	
		≥59	**Good**	

Table 3.2. Conservation status evaluation grid

The conservation status must be based on species composition, structure and ecosystem functionality (Bioret, 2002). Vegetation is an efficient integrator of local conditions like geomorphological changes and anthropogenic factors. Our conservation status assessment uses indicators based on vegetation and habitat structure and dynamics. Evaluation methods based on natural vegetation must include the appreciation of the presence/absence of specific vegetation types; the abundance of species and of characteristics species in each vegetation type; the occurrence of vegetation types in typical positions within the dune system (Araujo et al., 2002).

The habitat continuity corresponds to the presence of the community along the dune system and the surface occupied by the community. Vegetation cover is specific to every habitat (the recover of the annual vegetation of drift lines and embryonic dune is naturally low and may be hard to evaluate, therefore it is not included in the grid). The cover must take into account the path network density.

Habitat specificity marks out the physiological characteristics (proportion of characteristic species and non-native species). Human impact weakens the relationship between the vegetation and site conditions; the proportion of ruderal species reveals an anthropogenic influence by unspecialised man dispersion or nitrate supplies. Too high species diversity reveals the presence of ruderal or invasive species, but too low diversity corresponds to a deteriorated community (Richard, 2002). Habitat dynamics represent a good indicator of the global system: the relative stability of the different habitat surface shows a positive trend (Richard, 2002). The presence of an embryonic dune is an interesting indicator of coastal dynamics, it reveals a positive sediment budget and a good management of the beach (Williams et al., 2001; Favennec, 2002). Regressive dynamics can be detected on vegetation, for example: the absence or the reduced vitality of *Ammophila arenaria* in the mobile dune shows insufficient sandy deposits and a regressive dynamic (Corre, 1971). Partial disruption of the dunes changes the ecological conditions: embryonic and mobile dune vegetation types can be atypically found in interior dunes, indicating a regressive dynamic of the fixed dune (Araujo et al., 2002). The naturally is the opposite of artificialisation; it is the degree of interactions with human activities (Bioret, 2002).

We could also add a morphological aspect. With aerial photography, we can evaluate/quantify the shoreline evolution. The dune height and slope could be evaluated using Global Positioning System (GPS) and compared between several years.

The evaluation method we developed helps us to standardise studies and also to establish priority intervention between different settings. It is not a definitive tool. It may evolve and be improved. This assessment tool will be used for dune monitoring, in particular to evaluate the efficiency of management measures.

III.2. SALT MARSHES

III.2.1 LITERATURE REVIEW

[Based on text written for OD2.1]

Salt marshes can be effective as coastal defenses in two ways: *i)* in wave exposed areas, marsh vegetation can be capable of dissipating over 90% of incident wave energy over tens of meters (Möller, 2006) and *ii)* in more sheltered up-stream estuarine areas, the marsh offers water storage volume during spring tides or high river discharge. An important aspect in wave attenuation is that the root system of the marsh vegetation stabilizes the sediment against wave attacks (Barbier et al., 2008). The efficiency of hydrodynamic attenuation varies with plant community structure. Tall and rigid vegetation are more efficient dissipating wave energy than short and flexible species (Boorman, 1999; Bouma et al., 2005; Cooper, 2005), but this also depends on the density of vegetation canopy (Bouma et al., 2010). The reduction in flow energy stimulates sedimentation and allows a majority of salt marshes to keep pace with sea level rise (Asano 2006; Kirwan & Temmerman, 2009; Morris et al., 2002). The latter does however not resolve how some marshes have to cope with coastal squeeze as may arise due to sea-level rise (van der Wal & Pye, 2004).

The coastal defense value of salt marshes can be severely compromised by lateral marsh erosion, as a sufficient depth of marsh perpendicular to the sea is required to significantly reduce wave energy (Möller, 2006). Acknowledgements of such lateral dynamics for several wetland systems have given rise to important recent policy (e.g., 'making space for water' in the UK (DEFRA, 2004). However, much remains unknown about the effects of marsh-vegetation traits, as well as the importance and interactions between environmental parameters, such as relative exposure, sediment load in water, nutrient input, and tidal volume. Management-decisions and the consequences for the coastal defense-value of marshes can currently not yet be based on full scientific understanding.

Economic value coastal protection by salt marshes

Assets worth ~£120billion are at risk from flooding and coastal erosion in England and Wales (DEFRA, 2004). The UK governmental budget for managing flooding risk and coastal erosion for 2009/10 is £800 mill (DEFRA, 2004). The 1990 breach of the seawall in Towyn near Rhyl in North Wales cost in excess of £50 million and was blamed on a lack of natural protection of the foreshore (HR Wallingford, 2003). An estimated UK cost saving of £0.5 million/ha from coastal protection, principally by marshes (King & Lester, 1995), is widely thought to be conservative and out of date, and does not adequately reflect that the defense function itself varies under varying hydrodynamic (e.g. incident wave energy and water depth) conditions. We expect this UK example to be similar in other European salt marsh locations, except for low-lying countries like the Netherlands, where flooding impacts are bigger and thus damage will be higher. In more general terms, Costanza et al. (1997) valued disturbance regulation by global salt marshes and mangroves as $1,800/ha/yr, which is much higher than the value of food and raw material extraction ($466/ha/yr and $162/ha/yr, respectively). Thus salt marshes need to be considered as highly valuable ecosystems.

Performance in coastal protection

In this section, we highlight important studies showing that salt marshes contribute to coastal protection by attenuating wave energy. To our knowledge, Knutson et al. (1982) performed the first field study to quantify wave damping in smooth cordgrass (*Spartina alterniflora)* marshes (Chesapeake Bay, Virginia, USA). Wave height and the plant characteristics were measured at eight monitoring sites, and wave attenuation measurements were used to test an empirical model based on the approximation of expressing marshes as an array of vertical cylinders and the estimation of the drag coefficient. The *Spartina alterniflora* marshes report stem densities of 179–346 stems/m^2 and plant heights of 0.35–0.59 cm. It was shown that the reduction of wave energy in *S. alterniflora* marshes can decrease incoming wave energy by approximately 26% per meter of vegetation.

Möller et al. (1999) estimated wave attenuation over salt marshes in North Norfolk, England. The measurements of wind waves at three locations on a sand flat to salt marsh cross-shore transect showed that wave energy dissipation rates over the salt marsh were significantly higher (at an average of 82%) than over the sand flat (at an average of 29%). The average reduction of wave heights over 200 m was found to be ~63%. Also a numerical model based on theoretical energy dissipation was applied, showing that the main reason for wave attenuation is the bottom friction caused by the marshes as expressed by a bottom friction coefficient. Later measurements by Möller (2006) showed that vegetation type and density did not have a significant direct effect on wave attenuation, but modified the process of wave transformation. Based on the results at two locations on the east coast of England that are characterised by *Spartina spp*. dominated canopy (15-26 cm tall), the relative incident wave height (wave height/water depth) was identified to be the dominant factor on wave attenuation up to a threshold value of 0.55, beyond which wave attenuation showed no significant further increase. As flume experiments showed that, at least for at shallow water depths, plant traits like shoot stiffness and standing biomass have a dominant effect on wave attenuation (Bouma et al., 2005; 2010), this aspect requires further analyses within THESEUS.

Wave and current measurements done by Callaghan et al. (2010) on four salt marshes with contrasting wave exposure in the Scheldt estuary, revealed that sediment dynamics in front of the marsh plays an important role in the lateral development of a marsh, and the onset of cliff erosion. Moreover, the bathymetry in front of the marsh determines to a large extent how much wave exposure is effectively transferred to the leading edge of the pioneer vegetation and eroding cliff of the salt marsh. Hence wave attenuation over the foreshores is a topic we will emphasize within the THESEUS project.

III.2.2. MEASUREMENTS WITHIN THESEUS

Finalized measurements

Wave attenuation by salt marshes has been studied intensively in the Yangtze Delta, China (collaboration SKLEC & NIOO-KNAW). Wave attenuation measurements in the Yangtze Delta are interesting, regarding the large incoming waves. Here, waves of over a metre in height can enter the marsh vegetation during windy and stormy conditions. Measurements showed that the invasive pioneer species *Spartina alterniflora* had a major (positive) effect on wave attenuation and, therefore, increased the coastal defense value of the marshes considerably (Yang et al. submitted). Comparing wave attenuation measurements between the invasive tall and stiff pioneer species *Spartina alterniflora* and the native, short and flexible *Scirpus mariqueter* shows that both species have a much stronger wave attenuation than the bare mudflat, but that both species differ a lot (Ysebaert et al. submitted).

Data are not shown in this public document, as this would hamper publication in a scientific journal. The manuscripts that have been submitted for publication are on request available to the EU-commission (contact SKLEC: e-mail SKLEC contact; contact NIOO-KNAW: t.bouma@nioo.knaw.nl).

Ongoing & pending measurements

Within the THESEUS-project novel measurements on the coastal defense value of salt marshes, which expand on the previous work, have been carried out/are scheduled for the coming period. The majority of this work will focus on salt marshes of North Wales (UK) as a collaboration between Bangor University (BU) and NIOO-KNAW. The new series of measurements are specifically designed to address the questions that are raised by the study of Callaghan et al. (2010). So, in the new study the relative importance of the bathymetry in front of the salt marsh and the effects of vegetation on the landform itself are studied. The study is focused in North Wales, as this location offers comparison between a wide range of geomorphologic and hydrodynamic marsh configurations, in addition to the locations studied in the Scheldt and Yangtze estuaries. The set-up is expected to generate important insights into the conditions over which salt marshes can occur and develop.

For the campaign in North-Wales 7 locations for wave exposure and foreland measurements are selected (see fig.3.3). In order of expected exposure to wave energy the locations are:

1. **Y Foryd** – a very sheltered and expanding marsh. The marsh is muddy with a large pioneer zone of *Spartina anglica* monoculture.
2. **Maltraeth** (Cefni) – sheltered, may be expanding. Sandy, small pioneer zone of *Spartina a.* and *Puccenellia m*., dominant low-mid marsh.
3. **Morfa Harlech** – moderately sheltered (not known if expanding or stable). Sandy, banked, strong tidal action at cliff, some areas with, some without a clear pioneer zone. The higher parts of this marsh are grazed by cattle.
4. **Sunderland** marsh (Morecambe Bay) – moderately exposed. Sandy, 50 cm bank, tide and wave action.
5. **Morfa Aber**– moderately exposed to exposed; some stretches eroding, some expanding. Sandy, some pebbles, some mud. Pioneer or low bank (30 cm), with some sandflat patches of *Spartina.*

6. **Red Wharf Bay** – Exposed. Some areas are eroding, some expanding. Sandy, pebbles tide and wave driven.
7. **Gronant** – Very exposed. This location seems to erode. Sandy, small *Spartina* patches only.

Two symmetrical transects consisting of 7 wave sensors each, will be deployed for 48 hour to get insight into the relative importance of foreland of the marsh and the marsh itself on wave attenuation at each location. Because of the short deployments we expect that it will be possible to derive data on a large number of locations.

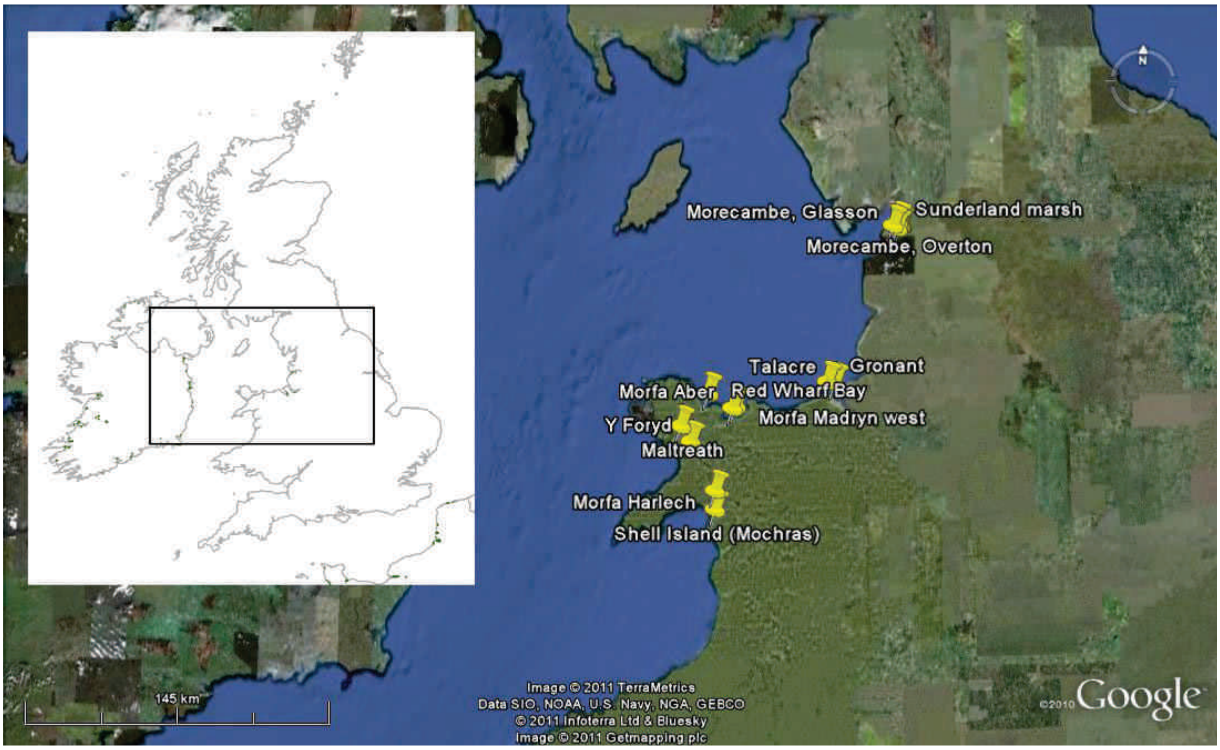


Figure 3.3. Location of potential salt-marshes for wave attenuation measurements.

The start of the measurements has been seriously delayed by delays in the delivery of the wave sensors, and technical difficulties with battery banks. This made it impossible to carry out the measurements that were scheduled for December to February 2010. As we can only do measurements on wave attenuation if weather conditions are stormy (>5Bft), these measurements have now been postponed to the fall and winter of 2011.

III.3. BIOGENIC REEFS

III.3.1 LITERATURE REVIEW

Biogenic reefs (including mussel beds and worm reefs) have been given many names in the past: “ecosystem engineers”, “foundation species”, “bioengineers” (Jones et al., 1997; Reise, 2002, 2005; Commito et al., 2005; Dubois et al., 2006; van Leewen et al., 2010). The majority of research on worm reefs and mussel beds as biocontructors has focussed on morphology of the structures and their function in enhancing biodiversity by habitat creation (e.g. Dubois et al. 2002; O’Connor and Crowe 2007). Few studies have focussed specifically on the geomorphological contributions of bioconstructions themselves, particularly in relation to their potential roles in natural coastal defence. Hence, there is much need for research on natural reefs both in terms of their intrinsic functions such as growth, development, resistance and resilience as well as their importance in larger scale ecosystem functioning. This research could be used to help improve our understanding of systems responses to large-scale environmental changes and, over time, to develop predictive models of responses to predicted sea-level rise and global climate change.

Sabellaria spinulosa

In gregarious aggregations, *Sabellaria spinulosa* tubes form a rigid structure (Hendrick & Foster-Smith, 2006). In most parts of its geographic range it does not form reefs but is solitary or found in small groups, encrusting pebbles, shell, kelp holdfasts and bedrock. When there are sufficient densities, reefs do form, and trap sand, detritus, and faecal material within the framework. Reefs are usually defined as structures that emerge from the seabed, and can be up to 60 cm high and can extending over several hectares. Reefs can persist in an area for many years although individual clumps may regularly form and disintegrate as sedimentary patterns change in an area (Jackson & Hiscock, 2008; Jones et al., 2000). Natural disturbance often causes damages to reefs as *S. spinulosa* is often found in in areas with quite high levels of natural sediment disturbance. In some areas, clumps of *S. spinulosa* may periodically break down and rebuild following storm events.

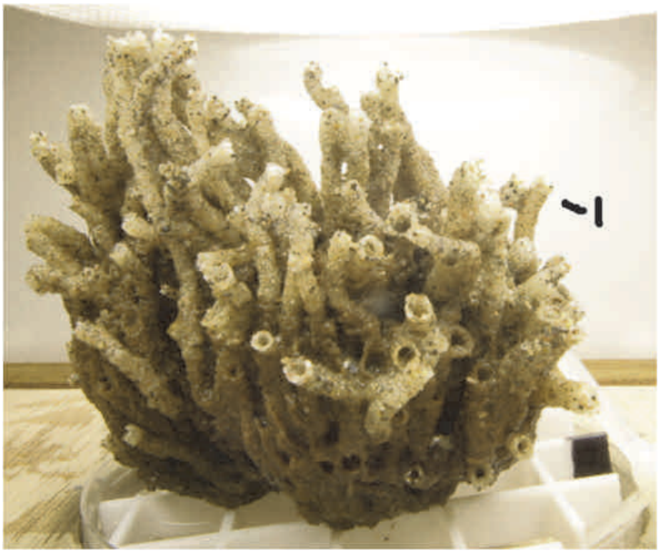


Figure 7.4: Colony of *S. spinulosa* collected from the Wash (Davies et al. 2009).

Sabellaria alveolata

This species constructs tubes from sand-sized (between 63 µm and 2 mm on the Wentworth Scale) mineral grains and shell debris. Sabellariidae also preferentially sort sand grains and accumulate heavy minerals (Fager, 1964; Gram, 1968; Multer and Milliman, 1967). *Sabellaria alveolata* is widely distributed throughout Europe. The geographic distribution of this species ranges from Cumbria, NW Britain to Mauritania (Crisp and Fischer-Piette, 1959; Gruet and Bodeur, 1995; Frost et al., 2005) and into the Mediterranean (Taramelli-Rivosecchi, 1961).

S. alveolata form two types of structures: *Veneers* and *reefs* (Fig. 3.5). Reefs are generally found at the lower level of the intertidal zone (Gruet, 1982) and can be up to 1.5 m in height. Thus, the reefs perform indirect and direct protectoral roles as they physically build structures that might influence local hydrodynamics or energy regimes, and preferentially store sediment that would otherwise be "loose" in the system, available to physically abrade shore platforms (Naylor and Viles, 2000; Naylor, 2001).

Figure 3.5. Example of *veneer* formation of *S. alveolata* in the UK (Photo: L. Firth).

Mytilus edulis

Large aggregations (beds) of *Mytilus edulis* are found in shallow marine environment along much of the coast of Europe. *Mytilus edulis* play a major role as bioconstructors forming beds that range in size from small clumps to large beds of several hectares (Dankers et al., 2001). In addition to this structural role in the environment, *M. edulis.* also modify their environment active filtration of seston including sediment and the subsequent formation of biodeposits (Flemming and Delafontaine, 1994; Oost, 1995) and passive influence of the rough and sediment retaining mussel bed (Widdows and Brinsley, 2002).

Examples include the use of mussels to reduce turbidity by biofiltration (Beukema and Cadée, 1996), which is thought to be beneficial for the reintroduction of sea grasses in the Wadden Sea (Van Katwijk, 2003). Furthermore, mussel beds can dissipate wave energy and thereby protect valuable salt marshes from erosion (Piazza et al., 2005; De Vries et al., 2007). Extra deposition of fine sediments in these areas by a reduction of flow velocities or fixation as (pseudo-) fecal matter is also thought to increase the resilience of salt marshes (Van Leewen et al., 2010).

Mytilus edulis individuals are semi-infaunal, so they project above the sediment, creating an irregular surface topography (Commito and Rusignuolo, 2000). This surface complexity alters water movement over the bottom, producing boundary layer flow regimes that affect the delivery and resuspension of sediment particles in mussel beds (reviewed in Commito and Dankers, 2001; Widdows and Brinsley, 2002; Widdows et al., 2002; Gutierrez et al., 2003).

Modiolus modiolus

Modiolus modiolus is found fully or partially buried in sediments (both soft and coarse), or attached to hard substrata using byssus threads. Individuals commonly aggregate into clumps and are capable, under suitable conditions, of forming extensive beds or reefs. Whilst individual horse mussels are found throughout most British coasts, extensive reef formations are rarer (defined as having a coverage of 30% or more). They occur through the accumulation of shell and faecal mud deposits and have been shown to support elevated local biodiversity (Holt et al., 1998).

III.3.2. MEASUREMENTS WITHIN THESEUS

Although biogenic reefs often are found in the lower intertidal and subtidal areas it is expected that these habitats still can contribute to a sustainable coastal defense due to cumulative effects and stabilization of landforms (Koch et al., 2009). Therefore, it is important to get more insight into the efficacy of wave attenuation by biogenic habitats under natural conditions. Within the THESEUS-project novel measurements on the coastal defense value of diverse biogenic habitats have been carried out or are scheduled for the coming period:

1. Mussels
    a. Waddensea (NIOO-KNAW, are scheduled for autumn/winter 2011)
2. Oysters
    a. Eastern Scheldt - Neeltje Jans (NIOO-KNAW, measurements over an oyster bed are done in January/February 2011. However, data have to be analyzed. Additional measurements in the Wadden sea are scheduled for autumn/winter 2011)
    b. Wadden sea (NIOO-KNAW are scheduled for autumn/winter 2011)
3. Seagrasses
    a. No measurements in WT3.1, but flume experiments in WP2
4. *Sabellaria*
    a. Sediment utilization by *Sabellaria alveolata* (BU)
    Records made since August 2010 to March 2011 of sediment utilized by S. alveolata from three sites along the coast of North Wales and the Lleŷn Peninsula. Laboratory experiments are planned for summer 2011 into the sediment utilization and erosion of *S. alveolata* under different sedimentary regimes.
    b. Long-term community trends (BU)
    Long term monitoring of three sites in North Wales has been undertaken since summer 2010 and analysis is currently ongoing. This will be supplemented by more detailed observations and aerial photography in autumn 2011.
    c. Wave attenuation measurements (BU)
    Scheduled for autumn/winter 2011, and will utilize two point measurements of wave height over whole *S. alveolata* reef beds. Three sites will be studied including Criccieth and Llanddulas, which contain some of the best developed reefs in the UK. This will be coupled with longer term dissolution experiments to measure the influence of small-scale reef fragmentation on hydrodynamics.

IV. MECHANISTIC DESCRIPTION OF DUNES, SALT MARSHES & BIOGENIC REEFS: A GENERAL EXPLANATION ON THE NATURAL ECOSYSTEM DYNAMICS, WHAT THREATENS THEM AND HOW THIS KNOWLEDGE IS RELEVANT TO MANAGEMENT

IV.1. DUNES

[Based on text written for ID1.4, OD2.1 & ID3.1]

PROCESSES AND MECHANISMS DRIVING NATURAL DYNAMICS & ECOSYSTEM DEVELOPMENT

Coastal sand dunes are aeolian landforms, found along the majority of the world's coast. This ecosystem located at the spatial transition between terrestrial and marine environments, can be found in coastal areas where a supply of sand-sized material (within the size range 0.2-2.0 mm) is available to be transported by winds. The coastal dune system is composed of the 3 compartments: the submerged beach, the emerged beach and the dune. These 3 compartments under permanent exchanges must be considered as a whole. Coastal morphodynamic variability is caused by a variety of factors ranging from climate and climate variability, relative sea level, sediment supply, vegetation, and coastal dynamics at global, regional, and local scales. Due to these factors, the coastal zone is a highly dynamic environment at temporal scales ranging from wind bursts and wave breaking (seconds to minutes), to storm and growing season variability (days to seasons), to interannual climate variability and sea level fluctuations (years to centuries, etc). The genesis of an aeolian dune is divided in three phases (Clemmensen et al., 2001): (1) dune field formation; (2) accumulation of sediment deposits; and (3) preservation of the deposited sediments. In order for coastal dune formation to begin, there must be both adequate sediment availability and sufficient wind energy capable of transporting this sand landward (Aagard et al., 2007). After the initial formation of the dune field deposits, accumulation of sand occurs when the influx of sediment is greater than the losses, creating a positive sediment budget. Finally, the third phase of preservation of sediments occurs when the dune system is stabilized either by the ground-water table rising or the growth of vegetation (Clemmensen et al., 2001), or a change in other climatic factors (Tsoar, 2005). Vegetation is necessary to trap the sand in order to make for dune growing, but also to stabilize the ground. The pioneer species, by this action, will facilitate the establishment of other species (less tolerant of salinity, wind…), increasing biodiversity richness (flora and fauna). A variety of factors affect the availability of sediment for dune formation, including changes in sea level, changes in sediment transport from continental and oceanic sources, and the presence of vegetation, as well as the impacts of human activities. In addition, the variability of the wind, both in the direction and magnitude, can play an important role in the mobilization and landward transport of sediment. The interactions between all of these forcing factors produce a variety of different environments in which dune formation may occur.

VULNERABILITY & THREATS

During the last thirty years, almost 75% of Mediterranean coastal dunes have been damaged or destroyed, principally by tourism consequences (Géhu, 1985; Salman & Strating, 1992; in Van Der Meulen & Salman, 1993). There are different kinds of destruction causes. First, the natural events which are erosion by storms or/and sea level rise, overwash, and sea flooding events. The vulnerability of coastal dunes to flooding depends on the characteristics of the dune system itself: height, width, conservation status etc. It also depends on the intensity and impact of the event (e.g. sea level rise, storm intensity). The taller dunes are more resistant to flooding but possibly more susceptible to erosion while the shorter dunes might be more vulnerable to flooding. In the next century, climate change will lead to a rise of mean sea-level, a likely increase of storms intensity and frequency and a more contrasted distribution of wetness between winter and summer (GIECC 2001, 2007, in Vinchon et al., 2008). These changes will modify the coastal erosion and sea-flooding hazards. Dune dynamics are driven by naturally occurring disturbances, which can be both common and recurrent. However when these disturbances increase in intensity or frequency or are removed there can be substantial alterations in community dynamics (Psuty and Martinez, 2004). Dunes are thought to be fragile because only a slight disruption (either natural or human induced) may lead to change and long-term progressive alteration (Carter, 1988) and their natural diversity might be compromised easily.

For thousands of years, human activities have been impacting the coastal environment of the Mediterranean Basin through agriculture, husbandry and the deliberate use of fire. In recent decades, tourism has caused important damages on coastal landscapes with the urbanization of the coast, the increase of summer visitors, and the introduction of invasive or exotic species. The most heavily affected habitats are the sandy coastal systems and coastal dunes in particular (Tzatzanis et al., 2003). The pedestrian and motorized pathways all over dunes lead to vegetation destruction and therefore enhanced weathering and erosion (Moulis and Barbel, 1999). Waste deposits and invasive species introduction are also destruction factors. The potential for dune recovery is dependent on the sediment supply in each area and on the intensity of human impact. Dune plants are especially sensitive to disturbance and are heavily affected by humans. Without dune plants, the integrity and preservation of a stable dune complex cannot exist. Anthropogenic impacts combined with the natural regression process of the coastline induce the acceleration of the destruction of the dune vegetation (Araujo et al., 2002) ultimately leading to to dune destruction.

KEY PROCESSES TO FOCUS ON FOR MAINTAINING ECOSYSTEMS INTEGRITY

Damaged coastlines are not attractive locations for tourism or leisure. Dune system vulnerability is defined as a set of conditions producing an acceleration of the erosion rhythm and system degradation. It is really important to not block natural processes which could destroy the system (Bodéré et al., 1991) and to take into consideration all the dune system with beach and foreshore. The natural dune-rebuilding process can take several years, and it may be desirable to rebuilding a storm-eroded dune quicker than natural processes (O'Connell, 2008). Dune damaging accelerates sand transit towards inland and then this sand cannot nourish the beach anymore (Pinot, 1998).

There are different ways to protect or restore dunes. Firstly, the protection of wildlife is important because fauna and flora are an integral part of dune system: vegetation stabilizes sand whereas fauna control plants growing and interactions. Sand dunes provide unique wildlife habitat. We must limit the trampling of visitors by paths and beach access setting up with fences. People walk through the dune because it is difficult to go on the sand. As they are looking for a hard surface to walk, we must provide them one. A marked pathway (by fences and why not with educative panels) is already dissuasion.

In case of dune landscape restoration, totally or partially destroyed, it could be necessary to stimulate natural vegetation regeneration by planting indigenous species, and preferably plants characteristic of the first step of dune colonisation: builder sand plants like *Ammophila arenaria,* or *Elymus fractus* because they permit sand stabilization and input. Moreover, it is fundamental to avoid invasive species and to limit them when there are already here.

The protection of implanted root native vegetation against wind erosion with weed permeable fences ("ganivelles") and biodegradable geotextiles (Heurtefeux et al., 2007) is also a priority.

Another action of dune rehabilitation could be sand input to ensure dune system dynamics. The nourishment of the lowest part of the white dune will restore a homogeneous altimetry of the dune barriers in order to make it less sensitive to natural aggressions (waves and marine wind), and to limit the risk of marine submersion. This must be adapted to the morphology of the dune with consideration for the sensitivity of the natural environment (Heurtefeux et al., 2007).

CURRENT MANAGEMENT PRACTISES

According to Heurtefeux et al. (in press), there are different coastal management modes which have been used and are yet to be employed. In the last twenty years, four global approaches to manage coast have been developed (**Error! Reference source not found.**). First, there is the approach by holding the line. As presented above, it has persisted to be used. Traditionally, the goal is to protect developed area using hard structures (Klein et al., 2001). The "do the minimum" approach corresponds to the use of natural processes to reduce risks but permitting coast natural changes. Some of the techniques used with this approach attempt to limit rather than to stop coastal erosion and cliff's retreat. The "do nothing" approach, which is rare but can be found is one of the more famous cases of "do nothing" approach is the municipality of Happisburg, in the county of North Norfolk (UK). The storm waves reached the coast with important damages on the bottom of the cliff, the cliff fell with major impact on the houses which were totally destroyed. Do nothing is one of policy adopted when it is too late, when any decision has been thought before, when the cost benefit analysis shows than the defence front to the sea exceeds the value of the properties. Finally, the Managed Realignment (M.R.) approach is quite recent. Its definitions and its particularities will be presented below.

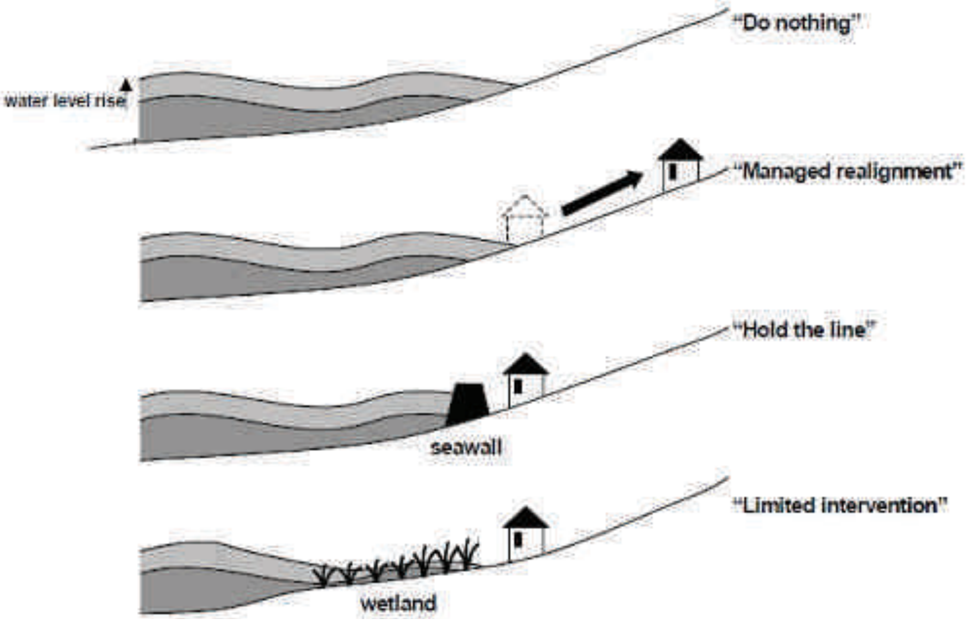


Figure 4.1.: Policy options for coastal management (European Commission, 2004; in Heurtefeux *et al.*, in press)

The M.R.'s aim is to avoid heavy structures to respect dune-beach system and its intrinsic transfers in order to not damage ecosystem functionalities. It's necessary to consider dune-beach system in its totality, and thus its natural capacity to return at an initial state after a perturbation.

There are many special features of Managed Realignment. One of these features is to move back the economic assets on the coast to the hinterland. It is also to create a new defence line behind the beach and facing the sea to restore natural areas and to create a buffer between the sea and the economic assets. Another feature is to avoid the construction of new economic assets in areas where they would be vulnerable (Heurtefeux *et al.*, in press).

IV.2. SALT MARSHES

[Based on text written for ID1.4, OD2.1 & ID3.1]

IV.2.1. PROCESSES AND MECHANISMS DRIVING NATURAL DYNAMICS & ECOSYSTEM DEVELOPMENT

Coastal areas, like estuaries, are high energy environments where organisms are exposed to hydrodynamic forces from waves and tidal currents. Ecosystem engineering species (Jones et al., 1997) play an important role in shaping the intertidal landscape (Temmerman et al., 2007; Weerman et al., 2010). Coastal vegetation, like salt marsh vegetation, are ecosystem engineers in that they can strongly attenuate hydrodynamic energy from tidal current and waves (Bouma et al., 2005, 2007, 2010). This has a positive effect on sediment accretion rates, and hence results in increased sediment elevation. In turn, increased sediment elevation stimulates the plant growth because inundation duration for the vegetation is shortened. This results in positive feedback between plant growth and sediment accretion. Implications of this feedback can be observed in the field in the form of dome shaped hummocks of cord-grass (*Spartina* spp.) that can be found on the mud-flats seaward of the salt marsh edge (Fig.4.2.), where the salt marsh is developing.

Feedbacks between hydrodynamic forces, sediment accretion and vegetation are key processes in shaping salt marshes (Temmerman et al., 2007; van Wesenbeeck et al., 2008). Locally the canopy of a vegetation stand can attenuates currents and waves which result in a net sedimentation. However, the same canopy also obstructs the flow, thereby diverting it and increasing flow velocities in the areas adjacent to the canopy because of conservation of mass and energy (Bouma et al., 2009). This biomechanical stress diversion can result in a negative feedback on vegetation settlement and growth at some distance from the canopy (van Wesenbeeck et al., 2008). However, as these kinds of feedbacks are density-dependent (Bouma et al., 2009), the strength of the negative feedbacks may vary with vegetation age, composition, or even the sediment type it is growing in (van Hulzen et al., 2007). So the overall outcome of these feedbacks may be dependent on the local context. Overall these feedbacks cause complex patterns of gullies and hummocks until eventually a mature marsh arises dissected by a complex drainage system (Kirwan & Murray 2007; Temmerman et al., 2007).

Many marshes are characterized by a cyclic nature, were marsh formation is followed by destruction. After a period of lateral extension, large scale lateral erosion of salt marshes can set in when the marsh edge becomes disturbed, a phenomenon often referred to as cliff erosion (see Fig. 4.2a; Allen 2000; Adam 2002). For example a disturbance from a storm surge, can initialize this erosion process by forming a steep slope. At the disturbed edge, sediment is more vulnerable to wave action and currents, so that once a cliff starts to erode, this process will not easily stop again. Thus the steep slope remains particularly vulnerable for waves and currents until it is protected by new marsh vegetation emerging in front of the cliff. The initiation of cliff erosion is intrinsic to natural temporal salt marsh dynamics (Allen 2000; van de Koppel et al., 2005). However, human activities, can contribute significantly to the severity of the cliff erosion (Allen 2000; Adam, 2002). For example, shipping traffic and dredging activities can increase exposure to currents and waves, thereby increasing the pace at which lateral erosion proceeds. Moreover, human induced activities may also take away the space for natural marsh recovery in front of the eroding cliff. The latter would result in the permanent loss of a marsh.

Loss of salt marsh habitat due to lateral erosion is a major problem across the world, especially in those locations where the marsh does not seem to recover. For example the marshes in the Venice Lagoon (Italy) laterally erode with 1.2-2.2 m yr^{-1} at their seaward edges (Day et al., 1998) and the estuaries of South-East England lose about 4,000 m^2 yr^{-1} of tidal marsh area due to erosion at the seaward edges and channel widening of creeks dissection the marsh (Hughes & Paramor 2004). However, the main drivers of salt marsh erosion are still subject of debate (Wolters et al., 2005). Generally it is believed that human activities are responsible for increasing erosion (Allen 2000; Adam 2002; Wolters et al., 2005), and pollution, shipping and dredging are some of the proposed anthropogenic causes. In addition, climate change and sea level rise are receiving much attention as a cause of salt marsh disappearance. In addition to these extrinsic forcing factors, intrinsic biological processes are also proposed (Allen 2000; Wolters et al., 2005). For example, vegetation-sediment feedbacks (Allen 2000) and sediment destabilization by bioturbation and herbivory by worms (Hughes & Paramor 2004; van der Wal & Pye 2004) and geese (Dionne 1985) can also result in erosion of salt marshes. A fundamental understanding of the mechanisms that control cliff initiation and salt marsh re-establishment in front of a cliff is needed in order to protect and manage these highly dynamic salt marsh ecosystems.

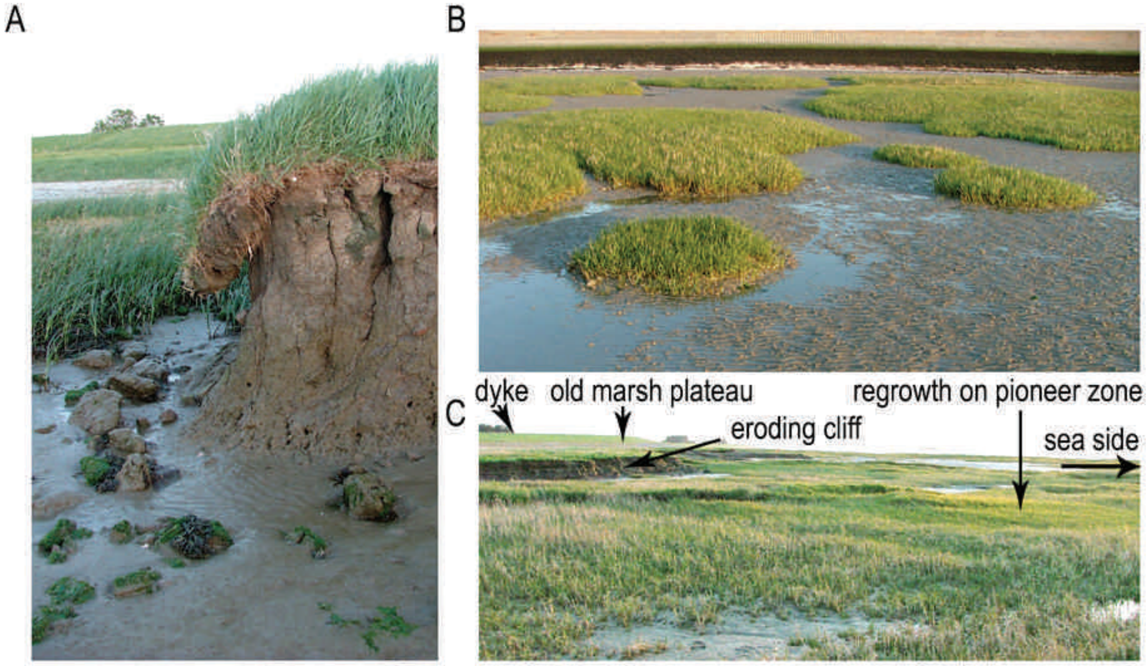


Figure 4.2 (A) Eroding cliff. (B) Patchy vegetation at pioneer zone of mudflat and saltmarsh interface. A dome-shaped patch is seen in front. (C) Salt marsh with eroding cliff separating the low marsh and pioneer zone. Pioneer vegetation (Spartina) has colonised the area below the eroding cliff (see also fig. 2.A). Photographs by J. van Belzen.

IV.1.2. VULNERABILITY & THREATS TO SALT MASRHES

Short-term effects of flooding and storms

SHORT-TERM FLOODING: VULNERABILITY OF MARSHES TO SALTWATER FLOODING

The salt-marsh community is well adapted to salinity due to regular tidal exposure to seawater. The great majority of salt-marshes are well drained and therefore at less risk to endured flooding. In comparison, the community of grazing-marshes is adapted to very dilute seawater and the habitat drainage is often slow. The potential impact of saltwater flooding is therefore more severe for grazing marshes than for salt marshes. Much of the evidence regarding the effect of seawater on coastal vegetation therefore relates to oligohaline/grazing marshes.

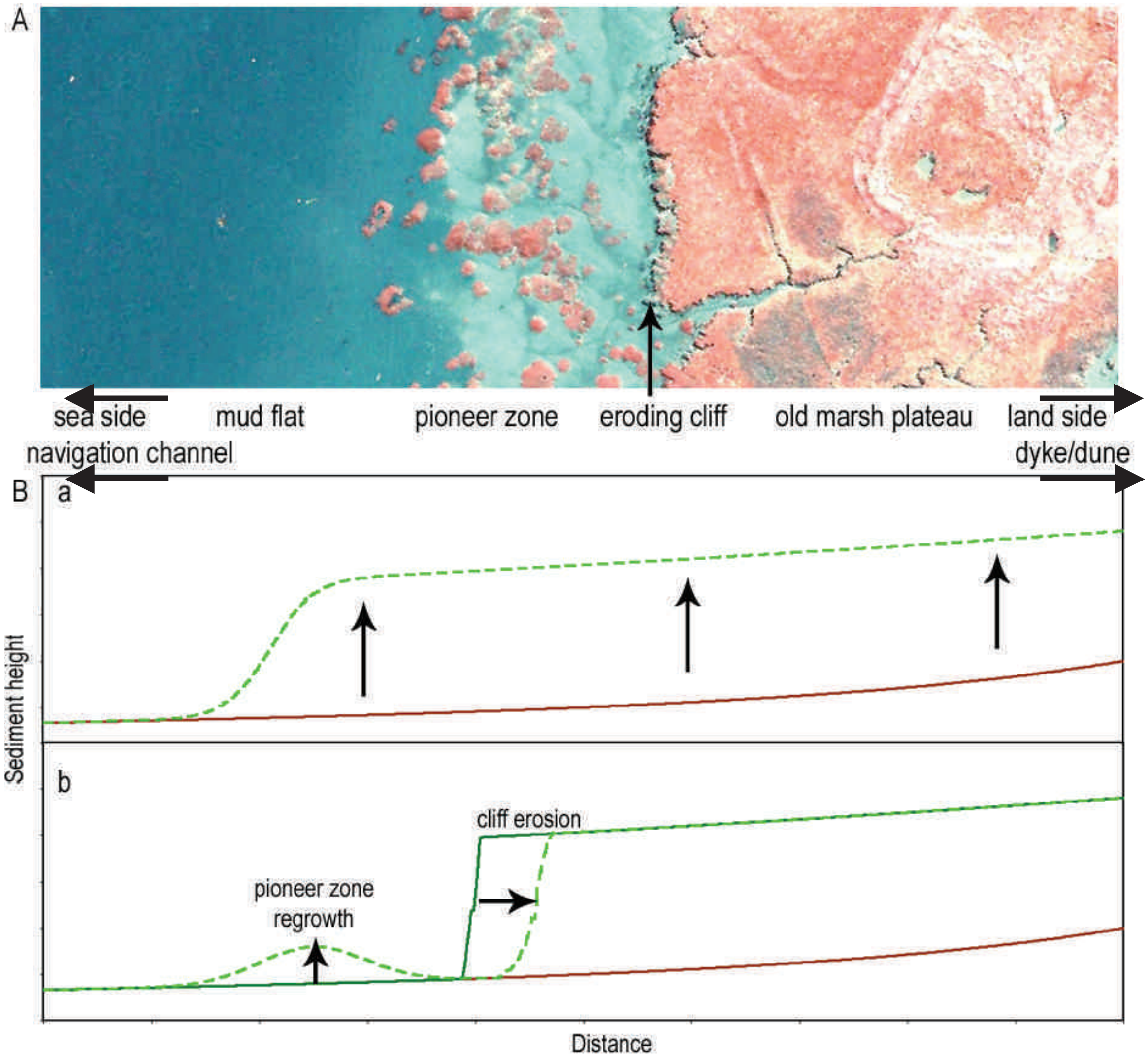


Figure 4.3. (A) Overview of saltmarsh by aerial photograph (RWS), comparable to situation (B.b) and see view from the ground in fig.1.1.C. (B) Cross-shore profile of salt-marsh dynamics of conceptual ecomorphological model, which mimics the development of the marsh in (A). Here, (—) is the initial bare mud-flat profile, (—) is vegetated marsh profile at beginning, and (- - -) is the final profile. First, (a) saltmarsh generation due to the positive feedback between vegetation and sedimentation. Second, (b) cliff erosion of old marsh and subsequent growth at the pioneer zone (after van de Koppel, et al., 2005).

SHORT-TERM FLOODING: Effect of flooding by saline water on salt marshes

A flooding event that originates from increased freshwater discharge, for instance due to heavy rainfall or ice melt in a catchment area, will result in a fresh-water pulse through downstream marshes. Unless freshwater flooding lingers for extensive periods, the impact on the vegetation of salt- and grazing-marshes will be short lived (Flynn et al., 1995; Grace and Ford 1996; Howard and Mendelssohn, 2000). The 'halophytic' (salt-tolerant) species that dominate salt- and grazing-marsh communities will not be harmed to short-term fresh water exposure, but their physiological and biochemical adaptations to cope with salinity stress makes them poorly competitive under fresh water conditions (Crain et al., 2004). The halophytic traits enable them to colonise saline environments (Pennings and Callaway 1992). Salinity exposure in the salt marsh, and consequently the inherent salt tolerance of the inhabitant community, does not necessarily linearly decline with shore level. Summer evaporation of seawater pools can leave concentrated deposits of salt on the high marsh where the habitat is infrequently flushed by the, leading to high levels of sediment salinity that exclude less halophytic species (Watson & Burne 2009). Paradoxically, increased frequency of seawater flushing by storms might dilute the accrued sediment salinity of such high marsh environments and alter the zonation of species. For instance, increased tidal flooding of an elevated marsh plain caused the normally very halophytic high marsh species to be replaced by salt-intolerant lower shore species (Watson and Burne 2009).

The severity of impact of salt water flowing is likely to depend on the natural salinity occurring at a specific location. Relatively brackish-marshes that are dominated by halophytic plants will see less changes to community composition than fresh water dominated grazing marshes and coastal flood plains (Brown et al., 1994). Increased flooding by salt water is most likely to have the greatest effect on the fresh-water adapted members of the marsh vegetation (Crain et al., 2004), which increase in dominance in the transitional and grazing-marsh above the tidal marks. Stormy conditions that result in a temporary increase in sea level and which brings in salt water pulses to coastal marsh systems therefore should have more potency for stressing the grazing-marsh community than the salt marsh community. Seawater flooding of a diked grazing-marsh, following a dike breaching, prevented most of the fresh water vegetation from developing in the following spring (Klein and Bateman 1998). Vegetation cover, species richness, recovery and re-establishment of an oligohaline marsh decreased following one month experimental exposures to increased salinity (from 0.5-5.0 salinity up to 12) (Howard and Mendelssohn 2000). In the longer term, the space left by dead vegetation is likely to be colonised by more salinity tolerant species, and thus grazing-marsh communities might come to resemble those of salt marshes (Doody 1982; Howard and Mendelssohn 2000).

Many coastal marsh plants are able to recover following temporary increases in salinity (Flynn et al., 1995; Grace and Ford 1996; Howard and Mendelssohn 2000). However, the potential for lasting changes to communities increases with the duration of flooding (Flynn et al., 1995; Howard and Mendelssohn 2000). Elevated salinity (from natural, 0.5-5, to 15) slowed vegetation recovery more in flooded than in drained soils (Flynn et al., 1995). Following temporary sea water floods the naturally slow drainage of grazing marshes make that this habitat remains immersed for longer periods than salt marshes. Grazing marshes are therefore more at risk to endured flooding, although indications are also that they are relatively resilient to exposure: if the water is brackish enough it may require months of immersion before significant impacts to vegetation cover occurs (e.g. Howard and Mendelssohn 2000). Brewer and Grace (1990) hypothesized that occasional storm-generated pulses of salt water moving into an oligohaline marsh would generate short-lived salinity gradients that, along with biotic interactions, regulated species distributions over longer terms. Sharpe and Badwin (2009) proposed that an unexpected peak in vegetation species richness in the transitional marsh arose because pulsed variation in salinity (0-5) prevented domination by fresh water or salt water species. Thus, pulsed salinity exposure might not necessarily depress vegetation diversity. Nevertheless, increased salt water flooding of grazing marshes is likely to drive succession towards more salt-tolerant vegetation, and increasing resemblance with salt marsh assemblages, although the empirical evidence for the rate of this transition is lacking (Nichols & Wilson 2001). The consequence of increased coastal flooding might therefore be gradual loss of grazing-marsh communities, in exchange for gain in area cover of salt marsh communities (Doody 1982; Klein and Bateman 1998; Nichols & Wilson 2001).

SHORT-TERM FLOODING: Interactions of salinity with other disturbances

It is important to caution against a wide interpretation that seawater flooding is a minimal risk to coastal marshes in general. The severity of seawater influence on grazing-marshes might depend much on whether the salinity is paralleled with other plant stressors and disturbances. Sharpe and Baldwin (2009) sampled plant diversity in a marsh in the United States, across a fresh (salinity 0.5) to mesohaline (5–18) salinity gradient. In an undisturbed marsh, richness in transition zone oligohaline marshes was as high as or higher than in tidal fresh water-marshes, but in an anthropogenically disturbed estuary plant species richness declined linearly with salinity increase. Experimental flooding by brackish (6-14) water had a greater effect on grazing-marsh community structure and biomass when the vegetation was also disturbed by leaf clipping (Baldwin and Mendelssohn 1998) or grazing (Gough & Grace 1998). In comparison, flooding did not affect species richness in the absence of such additional disturbance (Baldwin and Mendelssohn 1998). If the salinity and water regimes are permanently altered and/or the vegetation is destroyed by a combination of factors, the substrate might eventually subside. Substrate subsidence and associated increased water depth might prevent seed dispersal and germination of more flooding tolerant species, and thus hamper system recovery (McKee and Mendelssohn 1989). It is not known what the threshold frequency of flooding is, beyond which temporary events commence having such permanent effects.

SHORT-TERM: VULNERABILITY OF MARSHES TO STORM DAMAGE

Although flooding can induce some disturbances to vegetation composition of salt marshes, the threat to salt marshes from storm events is more likely to be induced from storm-associated damage. Wind-induced waves can destabilze sediments, initiate and propogate lateral cliff erosion, tear of plant material, and deposit of wrack and debris in marshes. The vulnerability of salt marshes is largely related to the effect of waves on sediment stability and lateral erosion. The evidence we present for effects of storm associated erosion nearly all originate from salt marshes; the literature on grazing-marsh damage from salt water erosion is scarce. Severe salt marsh erosion will undoubtedly lead to increased risk of sea water flooding and storm-associated damage for ajoined grazing-marshes, and might eventually drive a transition of grazing-marsh communities into salt marsh habitats.

SHORT-TERM STORM: Sediment destabilization and lateral erosion of salt marshes

Storm events can induce sediment stabilization and lateral erosion, which can have an important impact on the dynamics and functioning of salt marshes. Although lateral erosion is an intrinsic process for salt marshes, which is part of the natural cyclic behaviour, it generally gets initiated by a storm (Allen 2000; van de Koppel et al., 2005; Wolters et al., 2005). Such cyclic behaviour requires sufficient space for marshes to migrate landward. The space for the latter is nowadays diminished due to anthropogenic land use. Hence, lateral erosion of salt marshes has become a global threat, as it is unclear under which conditions an eroding marsh can re-establish in the limited available space. For example, the marshes in the Venice Lagoon (Italy) erode 1.2-2.2 m yr^{-1} at their seaward edges (Day et al., 1998) and estuaries of South-East England lose ~ 4,000 m^2 of tidal marsh per annum from erosion at the seaward edges and widening of creeks within the marsh (Hughes & Paramor 2004). In these locations, large areas of marsh are lost due to cliff erosion without little or no recovery of the vegetation.

It is clear that storms contribute significantly to the loss by lateral erosion. However, which factors are the main drivers initiating this erosion is still not clear (Wolters et al., 2005). Human activities can be in part responsible for increasing erosion rates through pollutant-driven diminishing of vegetation cover and/or by enhancing hydrodynamic energy reaching the marsh via ship waves and channel dredging (Allen 2000; Adam 2002; Wolters et al., 2005). There is thus a strong need for fundamental understanding of the cyclic functioning of salt marsh ecosystems in order to understand when disturbances by storms will start a natural cycle of rejuvenation versus when it cause the irreversible loss of a marsh and thus would benefit from protective measures.

Figure 4.4. Measuring of wave-breaking on a salt-marsh cliff. Insert: wave breaking during stormy conditions (photos by J. van Soelen, NIOO-KNAW).

Vulnerability of the saltmarsh to the initiation of cliff erosion will largely depend on the age of the marsh. Cliff erosion is hypothesized to be an inevitable and intrinsic consequence of the ecomorphological dynamics of saltmarshes (van de Koppel et al., 2005). That is, the capture of sediment by the vegetation leads to vertical salt-marsh growth, which in the long term makes the salt marsh susceptible to lateral erosion (Allen, 2000; van de Koppel et al., 2005). Both conceptual modelling and empirical evidence showed that a positive feedback between vegetation growth and sediment capture generates an increasingly steeper bank at the seaward edge of the marsh (van de Koppel et al., 2005). As a consequence, salt-marsh edges become more vulnerable to disturbance as they mature (see fig.4.2.B). This means that in the end, relatively small disturbances like from minor storm events or ship waves may induce the erosion. Data on sedimentation in salt-marshes, obtained from sediment core transects and spatiotemporal analysis of aerial photographs support this conceptual model (van de Koppel et al., 2005; van der Wal, et al., 2008), suggesting that, in a wide range of circumstances, lateral retreat due to cliff erosion will happen sooner or later.

Resilience of the marsh edge to erosion will depend on the interplay between vegetation composition and sediment dynamics. For instance, *Spartina* plants reduce cliff erosion more than *Limonium* plants do due to differences in their root system. As a consequence, a *Spartina* dominated marsh edge is likely to be less vulnerable to storm events (van Eerdt, 1985). However, consequences of plant community on the vulnerability and resilience of the marsh can be more complex. The above ground plant traits can have other effects on the vegetation-sediment interaction. For example, stiffness and density of the plant may affect sedimentation rates (Bouma et al. 2005, 2009, 2010). The overall effect of above and below plant traits on salt-marsh resilience/vulnerability remains largely unknown and subject to ongoing research.

Alongside cliff erosion, re-growth of pioneer vegetation on the cleared mudflat in front of the saltmarsh cliff ideally occurs, thereby rejuvenating the mature marsh (see Fig.2.B; van de Koppel, et al., 2005; van der Wal, et al., 2008). The pioneer vegetation determines the conditions for lateral retreat and gradually slows down erosion of the salt-marsh edge.

The establishment of pioneer vegetation is therefore of vital importance to the development and recovery of salt marshes (van de Koppel et al., 2005; van der Wal et al., 2008; Callaghan et al., 2010). This emphasises the importance of initial conditions for establishment and growth of vegetation, and the physical conditions that may constrain these intrinsic processes for salt marsh development (van der Wal et al., 2008). The factors limiting seedling establishment remains however poorly understood (Bouma et al., 2009). We hypothesize that sediment destabilization plays a critical role in the ability for pioneer establishment on a mudflat and, therefore, in the ability of the salt marsh to recover from lateral cliff erosion (Balke et al., submitted; Bouma et al., submitted; van Belzen et al., in prep a; Infantes et al., submitted).

SHORT-TERM STORM: Sediment and wrack depositioning on salt marshes

Deposition of wrack, debris or large amounts of sediment, associated with extreme flooding events, can have significant effects on salt marsh vegetation. Wrack depositioning may smother less hardy vegetation, leaving bare patches and opportunity for new colonisation by neighbouring species, or from dispersed seeds (Bertness and Ellison 1987; Tolley and Christian 1999). Large algal mats can have residence times of 3-4 months (Valiela and Rietsma 1995). While wrack depositioning may not be as significant in cover (Valiela and Rietsma 1995), small-scale alteration in species cover by algal wrack deposition does have the potential for wider effects on community diversity if the same spots are regularly covered by seaweed (van Hulsen et al., 2006). Sediment depositioning following storm flooding may be significant. Experimental flooding and sedimentation of seedbanks of an oligohaline marsh community showed that addition of 2 cm of sediment decreased plant density and germination of seedlings, suggesting that increases in sedimentation and relative sea level may reduce plant biodiversity (Peterson and Baldwin 2004). Sediment depositioning might be positive for subducting and nutrient starved marshes. Within a year after 3-8 cm sediment depositioning by Hurricane Katrina the vegetation of a high marsh had fully recovered and below-ground root growth had increased 10-fold (McKee and Cherry 2009).

While single factors may have limited effects on marshes, a collective of concurrent stressors are likely to generate significant impact on marsh communities. Wrack and sediment depositioning are often concurrent with other habitat stressors that might jointly influence marsh vegetation cover. Thus, while Tolley and Christian (1999) found little effect of sea water flooding on vegetation biomass, the simultaneous deposition of algal wrack greatly depressed plant cover and biomass, in some species irreversibly so.

Long-term effects due to climate change and sea level rise

Coastal squeeze, due to sea level rise, and erosion are primary threats to salt marshes across Europe. They can result in reduced coastal defense value and increased risk of flooding. Although sea level rise may pose serious threats to the survival of salt marshes, there is growing evidence that if sediment supply is sufficient, the vegetation-sedimentation feedback of marshes enables marshes to accrete vertically at the rate of the rising sea-level (Kirwan & Temmerman 2009). If suspended matter load is reduced by climate change or significant human alteration in a catchment area, vegetation-sedimentation feedbacks can become limited, affecting the potential of marshes to accrete (Kirwan & Temmerman 2009). As explained in the previous sections, lateral marsh erosion becomes over time a serious threat to salt marshes if seedling establishment in front of the marsh is not possible so that re-growth of the marsh is prevented. Many aspects that affect the cyclic dynamics of marshes are still not well understood. Important in maintaining the vegetation-sedimentation feedback is that the sedimentary conditions remain more or less the same.

Several managerial aspects are likely to compromise the capacity for marshes to persist and to protect the coast. Reduction in area by coastal squeeze will reduce the wave attenuation capacity, as the efficiency of energy reduction is strongly dependent on the depth of the marsh (Möller 2006). Whether this might have negative feedback on marsh accretion and accelerate area loss is not known. Effects of grazing of might also reduce the vegetation-sedimentation feedback by to reducing vegetation cover and height, thereby hampering the development of salt marshes (Kiehl et al., 2007). Finally, very little is known about the implications on salt marsh resilience from interactions between different environmental, climatic and managerial variables. Interactions between climate stressors (e.g. desiccation, irradiation), physical forcing (extreme flooding events, increased storminess) and environmental management (eutrophication, grazing, and managed retreat) will be a likely reality for many marshes. This is important because interactive stresses can be synergistic and cause shifts in ecosystem stable states, with consequent compromise of the natural services delivered (Scheffer et al., 2001; Scheffer et al., 2009). Our evaluation of the resilience of salt marshes to disturbance, including climate change, might for the time being still be somewhat naïve and informed by limited current research.

Forecasting the effects of sea-level rise at Chongming Dongtan nature reserve in the Yangtze Estuary

Located at the mouth of the Yangtze Estuary, the Chongming Dongtan nature reserve is extremely vulnerable to climate change and especially to accelerated sea-level rise. We use a variety of data from remote sensing, an in situ global positioning system (GPS), tidal gauges, nautical charts, geographic spatial analysis modeling and IPCC sea-level rise scenarios to forecast the potential impacts of increased sea level on the coastal wetland habitat at Chongming Dongtan Nature Reserve. The results indicate that around 40% of the intertidal zone of the nature reserve will be inundated by the year 2100 due to an estimated 0.88 m increase in sea level. In particular, the *Scirpus mariqueter* communities and bare tidal flats are more vulnerable to sea-level rise. The identification, mapping and statistical summary of environmental impacts of the projected sea-level rise at Chongming Dongtan Nature Reserve represents an important initial step for decision makers concerned with mitigation of the adverse impacts of sea-level rise. In this study, the inundation-based assessment was developed to inform policymakers, mangers and the public about the amount and spatial distribution of tidal wetland change as a result of sea-level rise. The results indicate the zones most vulnerable to sea-level rise at the Chongming Dongtan Nature Reserve is the *S. mariqueter* zone, the bare tidal flat zone and the tidal creeks, which are the most suitable habitats for migratory birds. A ~30% loss of the *S. mariqueter* marsh community by the year 2100 would eliminate a rich invertebrate food source and cause deterioration in the estuarine food web for migrating birds; such a loss could arise from human-induced stressors such as land reclamation, seawall constructions, overfishing and local pollution. As tidal marshes and flats submerge and decline in size and productivity, increased crowding in the remaining areas could lead to reductions in and eventually even exclusion of some local shorebird populations (Tian et al., 2010).

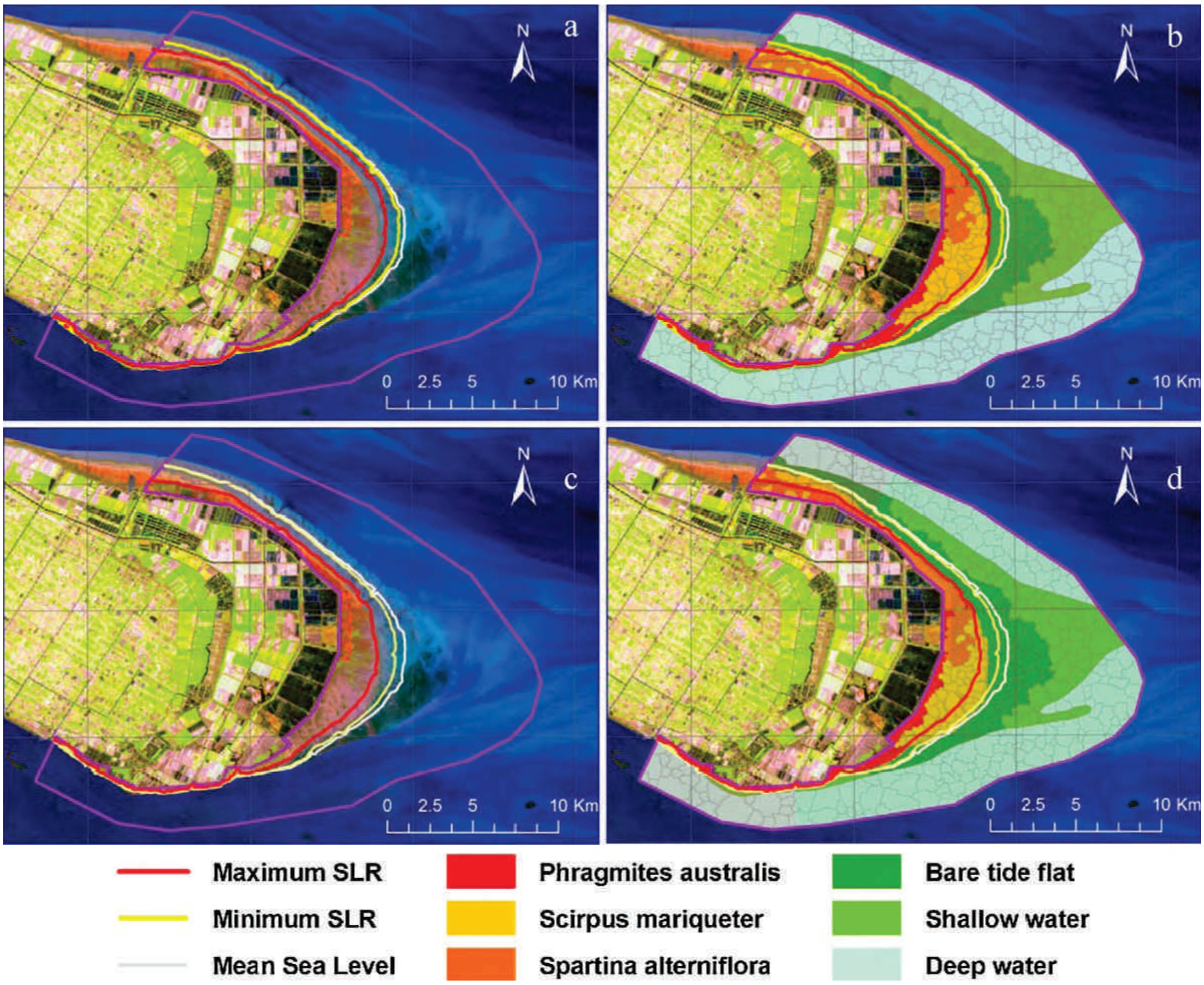


Figure 4.5. Impact of sea-level rise on tidal flat and tidal marsh complex by 2050 (a and b) and 2100 (c and d) year at Chongming Dongtan nature reserve.

IV.2.3. KEY PROCESSES TO FOCUS ON FOR MAINTAINING ECOSYSTEMS INTEGRITY

Effects of single disturbance events on marsh responses to long-term change

Single events, such as violent storms, normally have short-lived effects on the species composition and ecological functioning of salt-marshes (Flynn et al. 1995, Howard and Mendelssohn 2000, McKee and Cherry 2009), and are thus of reduced importance compared to long term persistent changes in environmental condition. Long-term processes of coastal squeeze with sea level rise and lateral erosion with increased storminess are considered to be the primary threats to salt- and grazing-marshes across Europe (Nichols and Wilson 2001). A single storm can push a marsh over the tipping point, shifting it from laterally expanding towards laterally eroding. If erosion persists, and the marsh cannot re-establish in front of the cliff, over time this will result in reduced coastal defence value and increased risk of flooding of adjacent terrestrial environment (e.g. grazing-marshes) (Klein and Bateman 2007).

IV.2.4. CURRENT MANAGEMENT PRACTICES

Making space for water

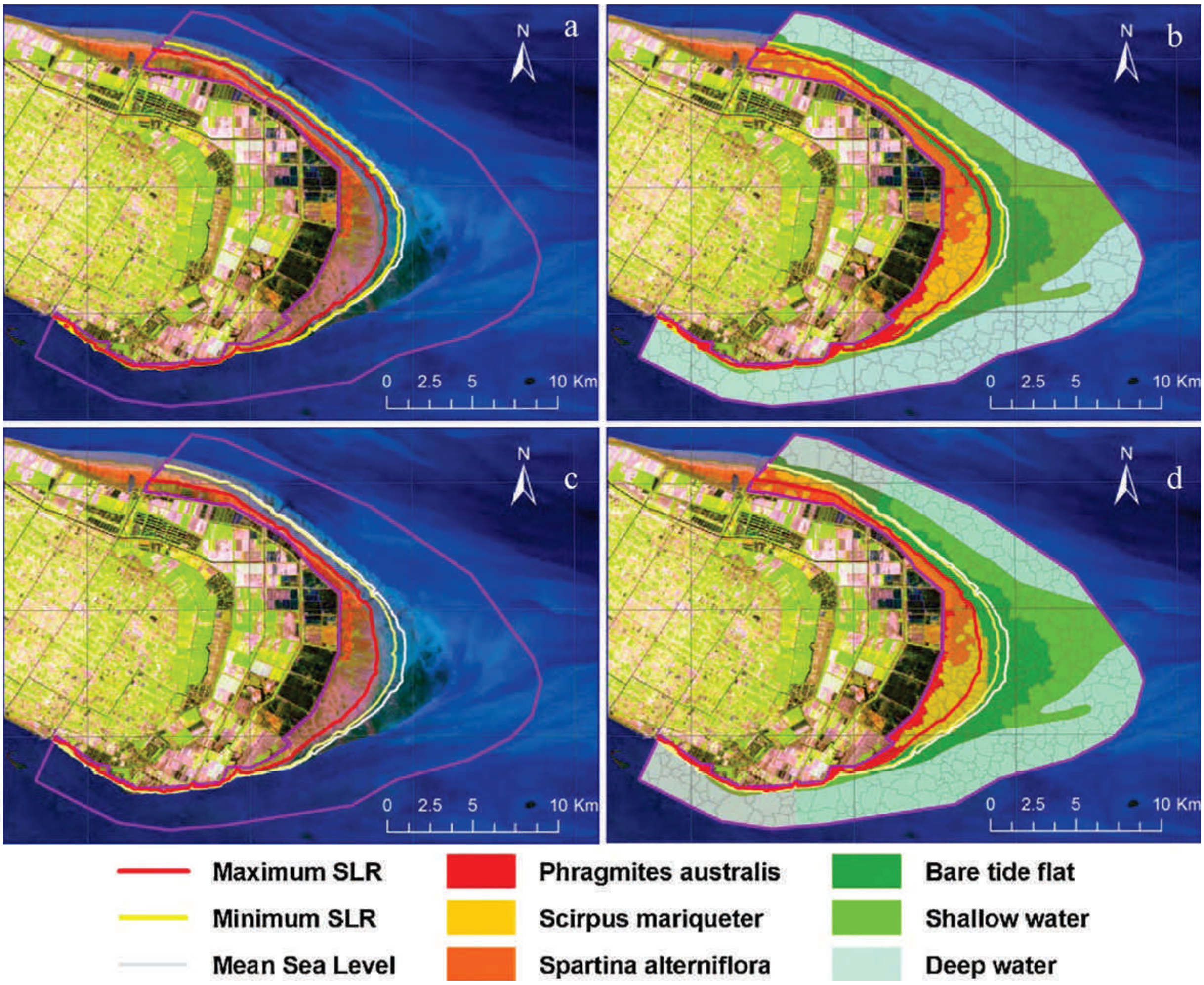
Currently, salt-marshes are managed extensively because for their acknowledged role in coastal protection. Many countries like e.g. the UK, the Netherlands, etc, have developed management schemes seeking to make space for water along river flood plains, estuarine and coastal areas (Bakker, et al., 2005; DEFRA, 2004). In this way, river run-off and occasional high sea water levels can be attenuated by the natural buffer and retention capacity of the landscape. For example, restoring the water storage volume in an estuary can reduce the tidal prism, smoothing the tidal amplitude, which reduces the risk of flooding in up-stream estuarine areas. Salt-marshes play an important part in this contemporary policy, because creating new marsh-land both increases tidal water storage in up-stream estuarine areas and wave attenuation of storm surges along exposed coast lines (Bakker, et al., 2005; Kiehl, et al., 2007).

Managed retreat/realignment and salt-marsh engineering

The current effort to restore marsh systems in Europe and elsewhere represents graphic evidence of the political and managerial value placed on the goods and services provided this ecosystem. The principle of 'managed realignment' and 'managed retreat' is one of allowing salt-marsh areas that were historically converted to alternative use for anthropogenic purposes (e.g. agricultural land or tourist development) to return to their natural state and area cover (Garbutt, et al., 2006). This can be done in a number of ways, but typically involves making a breach in the historically erected barrier (seawall, dike) rather than removing the whole structure. This approach reduces the cost involved, as well as the wave action depressing the development of the vegetation. Cost benefit analyses typically show a net advantage of managed realignment over other constructed defense options (Turner, et al., 2007). Full restoration of natural ecosystem function has met with some complications. The substrates and biodiversity of pristine salt marshes is often markedly different from an artificial or restored system, even 100 years after natural processes have been allowed to operate (Hazelden & Boorman, 2001). The implications of this on coastal protection by marshes are not known. The MOSE project of the Venice lagoon is an impressive example of large-scale engineering to create salt-marsh wetlands, largely for their role in dampening wave action and erosion within the lagoon (MOSE 2010).

Grazing management and coastal protection

There is evidence to suggest that grazing management could have particular relevance to the capacity of marshes for protecting the coast, although there has been little quantitative research on this subject (Bakker, et al., 2005). The vegetation has a key importance to coastal protection by marshes, through consolidation of the soil and by representing a structural hindrance to wash-over waves. Evidently, livestock has large potential for altering vegetation structure directly through feeding and indirectly by altering the condition for vegetation growth (Bakker, et al., 2005; Kiehl, et al., 2007). Feeding and defecation moderates vegetation structure-composition and above- and below-ground biomass production. Trampling and hoof holes lead to soil compaction and can cause saltpan formation (Vera, 2000). The potential for management of grazing regime to influence salt-marsh coastal protection potential is therefore high. Intense grazing modifies zonation patterns and transforms complex communities with woody species into homogenous lawns dominated by short flexible grass (Andresen, et al., 1990; Kiehl, et al., 2007), with an associated likely reduction in wave attenuation (Möller, 2006) and sedimentation rates (Andresen, et al., 1990). Grazing at low intensity increases vegetation patchiness and biodiversity due to selective grazing of palatable species (Bakker 1985, 1998; Kiehl et al 1996; Adler et al. 2001; Bouchard et al. 2003; Marriot et al. 2005). Patchiness may causes specific spatial patterns in turbulence and sedimentation (Boorman, 1999; van Wesenbeeck, et al., 2007), so that the sum effect of patchiness on marsh coastal protection is not known. Conversely, grazing pressure can lead to greater resource allocation to below-ground biomass (Pucheta, et al., 2004), thus reducing surface erosion and below ground contributions to an increase in marsh surface elevation.



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IV.3. BIOGENIC REEFS

[Based on text written for ID1.4, OD2.1 & ID3.1]

IV.3.1. PROCESSES AND MECHANISMS DRIVING NATURAL DYNAMICS & ECOSYSTEM DEVELOPMENT

In this section, the processes and mechanisms driving natural dynamics and ecosystem development of biogenic reefs are discussed for each group in turn: *Sabellaria spinulosa, Sabellaria alveolata, Mytilus* spp. and *Modiolus modiolus*.

Sabellaria spinulosa

Environmental Requirements

S. spinulosa is thought to require stable foundations on which to settle and establish a tube (Jackson 1977; Wood 1999; Chisholm & Kelley 2001) and is thus likely to favour substrata which include bedrock; boulders, cobbles, mixed substrata; and mixed sediment (Connor et al., 1997). Although it is assumed that a firm substratum is required for colony establishment, it has been suggested that the reef can be subsequently increased in extent by addition to the existing colony without the need for hard substratum (Holt et al., 1997). Many studies have reported extensive colonies in predominantly sandy areas (Warren & Sheldon 1967; Schäfer 1972; Warren 1973; Limpenny et al., 2010). Recent observations from The Wash, England show that *S. spinulosa* had 'seeded' on shell fragments predominantly from blue or horse mussels (Ian Reach, Natural England, pers. comm.).

As *S. spinulosa* is a sedentary species, it relies on wave and current action to bring food and wash away waste products (Kirtley 1992). Strong water movement is required for food provisions but is perhaps more important to raise sediment into suspension for tube building (Jones 1999). As a result, *S. spinulosa* colonies are typically located in areas of weak to moderately strong water flow (Jones et al., 2000). It also appears to favour locations around the edges of sand banks or areas where there are sand waves (Foster-Smith 2001a). *S. spinulosa* typically occurs subtidally in depths of a few metres to up to 40 m depth (Caspers 1950; George & Warwick 1985; Connor et al., 1997; Jessop & Stoutt 2006) but can occur in depths up to 600 m (Hartmann-Schröder 1971). *S. spinulosa* occasionally occurs in the lower intertidal zone (Jessop & Stoutt 2006).

Reproduction and Development

The fecundity and recruitment of *S. spinulosa* is known to be variable (e.g. Linke 1951; Wilson 1971; Michaelis 1978; George & Warwick 1985). The family Sabellariidae are broadcast spawners, reproducing sexually, resulting in larvae that drift passively in the plankton (Schäfer 1972; Eckelbarger 1978). The larvae can spend between a few weeks to several months in the plankton (Wilson 1929) before seeking appropriate conditions for settlement (Wilson 1968; Eckelbarger 1978). If conditions are unsuitable, the larvae are able to delay metamorphosis for several weeks. Physical factors alone have limited influence on settlement (Wilson 1968) and settlement and metamorphosis is strongly influenced by the tube cement of other sabellariids (Wilson 1968; 1970; Eckelbarger 1978; Jensen 1992). This mechanism ensures settlement in a suitable habitat and promotes the development of large colonies.

Despite only a few studies investigating the rates at which *S. spinulosa* can extend their dwelling tubes (Hendrick 2007 and Davies et al., 2009 being exceptions), it appears that sabellariid reefs develop quickly following successful settlement (Linke 1951; Vorberg 2000; Stewart et al., 2004; Braithwaite et al., 2006). Last et al. (2011) observed that tube extension rates to be highly variable and that they could grow up to 6 mm a day for several days when provided with an adequate sediment supply.

Little is known about the longevity of *S. spinulosa* colonies, but estimates for sabellariids estimate that they are expected to survive for 1-2 years (Kirtley 1966; McCarthy 2001; McCarthy et al., 2003), with some reports of longer life spans (Wilson 1974; George & Warwick 1985). It is likely that the age of an actual colony may greatly exceed the age of the oldest individuals. This is particularly likely as sabellariid larvae are stimulated to metamorphose by conspecific secretions, encouraging continuous succession of generations.

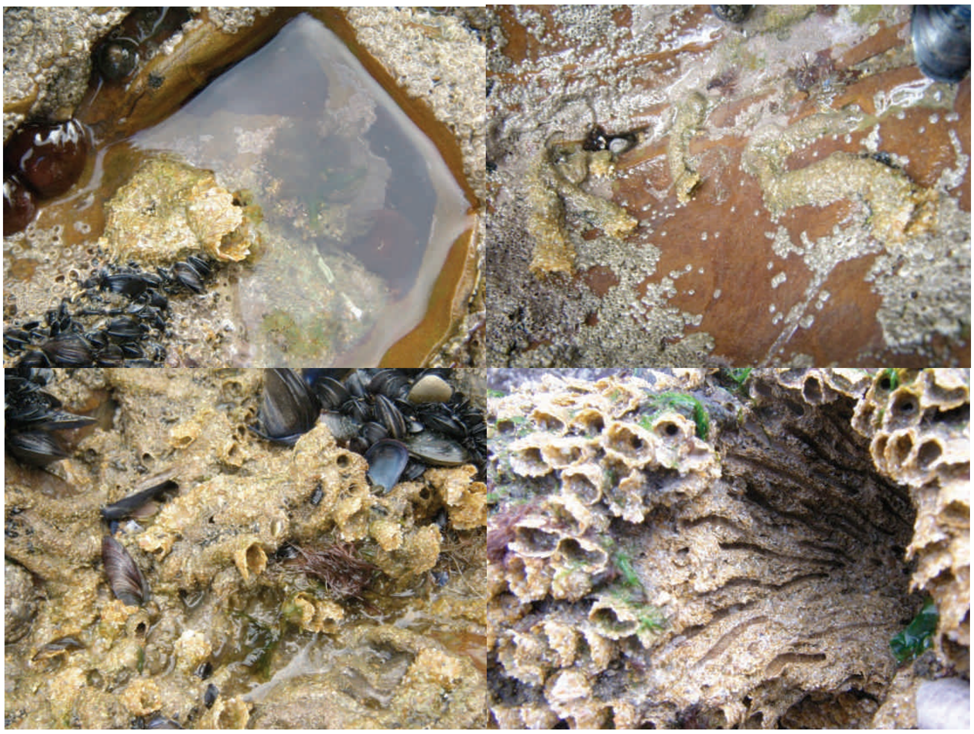

Figure 4.6. Images illustrating the various stages of development of *S. alveolata* reef at Bude, Cornwall (photos L. Firth).

Sabellaria alveolata

Environmental Requirements

S. alveolata generally requires hard substrata on which to develop, but that these must be in areas with a good supply of suspended coarse sediment for tube building. *S. alveolata* reefs are known to form on a range of substrata from pebble to bedrock (Cunningham et al., 1984). Reefs therefore commonly form on areas of rock or boulders surrounded by sand. Larsonneur (1984) noted that settlement of *S.*

alveolata was facilitated by the sand mason *Lanice conchilega* which can stabilise sand well enough to allow colonisation by *S. alveolata*. Settlement occurs mainly on existing colonies or their dead remains (Fig. 4.6).

Water movement of sufficient intensity to suspend coarse sand particles, making them available for building the worms tubes, is a prime requirement. Cunningham et al. (1984) note that this may consist of waves or currents. In many British localities such as the south west of England, much of Wales and the Cumbrian coast the former seem more important, but in others such as parts of the Severn Estuary tidal suspension is probably very important. However, *S. alveolata* is generally absent from very exposed peninsulas such as the Lleyn, Pembrokeshire and the extreme south west of Cornwall, which probably relates to the effect of water movement on recruitment (Cunningham et al., 1984).

S. alveolata reefs generally form on the lower third of the shoreline and are occasionally reported to exist in the shallow subtidal. The paucity of subtidal reefs could be due to lack of data and survey effort. Extensive shallow subtidal reefs have been reported in depths of up to 20 m in the Severn Estuary (Bamber & Irving 1997), whilst other patchy subtidal reefs are known from the Cumbrian and Northern Ireland Coasts (Perkins 1981 ; Erwin et al., 1990).

Reproduction and Development

It is thought that the larvae of *S. alveolata* spend anything from 6 weeks to 6 months in the plankton (Wilson 1968; Wilson 1971) in order to attain widespread dispersal. The most detailed work done on *S. alveolata* reproduction in the British Isles is that of Wilson in Cornwall (e.g. Wilson 1971). Wilson observed slight settlement in all months except July, but in 14 years of monitoring (1961 to 1975), Wilson (1976) observed only three heavy settlements, in 1966, 1970 and 1975. All occurred from September to November or December. Subsequent studies have revealed that the intensity of settlement is extremely variable, both temporally and spatially (Gruet 1982 ; Cunningham et al., 1984). Settlement occurs mainly on existing colonies or their dead remains; chemical stimulation seems to be involved, and this can come from *S. spinulosa* tubes as well as *S. alveolata* (Wilson 1971 ; Gruet 1982 ; Cunningham et al., 1984).

Mytilus spp.

Environmental Requirements

The widespread distribution of the *M. edulis* is a reflection of its tolerance of a wide range of environmental variables. Natural reefs typically occur on firm, mixed sediments in relatively wave sheltered estuaries and bays characterised by strong currents (Holt et al., 1998). In more exposed exposed areas, larger colonies are only able to develop on hard and stable substrata such as rock or large boulders (Seed 1969). Conversely, in sheltered environments large beds may develop on more sandy substrates (Roberts & McKenzie 1983).

Mussels produce byssal threads which anchor them to the substratum and each other, enabling large beds to develop. Mussels can grow in all but the most exposed conditions where their byssus threads can provide anchorage against wave action and water flow. As *M. edulis* is a sessile filter feeder, it requires sufficient water to flow to bring food and wash away waste. Larger beds require higher flow in order to provide sufficient food supply to high numbers of individuals. It is generally considered that this water movement is best provided by tidal currents rather than wave action, though the latter may also contribute in some areas (Holt et al., 1998).

M. edulis is tolerant of a wide range of salinities, being found in locations ranging from estuarine to fully marine but larger reefs typically occur within the lower third of the intertidal and in the mid to lower reaches of the estuary (Holt et al., 1998). *M. edulis* reefs do form subtidally and have been reported to occur at depths of 30 m (Ian Reach, Natural England, pers. comm.). The upper limits of *M. edulis* are thought to be set by temperature and desiccations stress (Seed & Suchanek 1992) in addition to reduced feeding (Widdows & Shick 1985). The lower limits are generally set by biological factors such as competition and predation with physical factors playing a secondary role (Holt et al., 1998).

Reproduction and Development

M. edulis fecundity and recruitment success is highly variable, both temporally and spatially. It can reproduce in its first year and can spawn throughout the year with a major spawning event usually occurring in the spring (Seed 1969). Larvae can survive in the plankton for between 2-4 weeks before metamorphosis, although this can be up to 6 months, depending on availability of food, suitable substrate and temperature (Holt et al., 1988). Settlement can be either a one-stage or a two-stage process. Some larvae can settle directly onto adult beds (McGrath *et al.*, 1988) or they can temporarily settle onto sublittoral filamentous substrata such as algae or hydroids before becoming detached, and eventually settling into an adult bed (Bayne 1964; Pulfrich 1996). It is thought that this may be a mechanism for reducing competition between very young and adult mussels, and/or to prevent filtration of the larvae by the adult mussels. Mc Grath et al. (1988) reported very large densities of settling spat in Ireland, but more commonly modest recruitment between the shells of adult mussels provides sufficient supply to maintain persistent beds (Holt et al., 1998). Conversely, heavy recruitment may not necessarily lead to the formation or maintenance of a dense bed or reef if predation or losses due to wave action are high.

M. edulis growth and production can be extremely high, particularly in sheltered or estuarine areas (Holt et al., 1998). It has been reported that *M. edulis* accounts for 20% of the the total macrobenthic production in the Wadden Sea (Beukema 1983), whilst Dare (1976) estimated the production by two year classes to be 2.5-3 times their maximum standing crop, with few mussels surviving beyond their third year. It is thought that the majority of mussels do not suvive beyond 3 years of age (Seed 1976), there are reports of individuals surviving beyond 15 years (Sukhotin et al., 2007).

Modiolus modiolus

Environmental Requirements

Despite typically occurring on hard substrata, *M. modiolus* beds and reefs are capable of forming on a variety of sedimentary bottoms ranging from muddy substrata in some sea lochs to quite coarse mixed sediments containing much stones and shell. Larvae can also settle on artificial substrates such as oil rigs and can form reefs on these structures. The byssus threads of adult *M. modiolus* provide a suitable substrate for attachment and protection from predators. Beds occurring infaunally can lack available byssus threads and thus limit recruitment (Holt & Shalla 1997) and the development of larger beds.

M. modiolus have a very wide depth distribution typically being found subtidally from a few metres depth right down to depths of 280 m (Schweinitz & Lutz 1976). Intertidal populations have occasionally been reported (Davenport & Kjosvik 1982), but these are thought to be limited by temperature and desiccation stress associated with aerial exposure (Coleman 1976; Davenport & Kjosvik 1982). The densest populations that are known as reef are found between 5 and 50 m in British waters (Holt et al., 1998), whilst infaunal reefs have been found at over 80 m in the Bay of Fundy (Wildish et al., 2009).

Reproduction and Development

M. modiolus is a long-lived species with individuals only reaching sexual maturity between 3 and 6 years of age. It is thought that this adaptation is in response to high predation on juvenile mussels, thereby channelling energetic resources towards growth in early life. As a result, *M. modiolus* exhibit rapid growth in the first few years of life, followed by much slower growth following sexual maturation (Anwar et al., 1990). *M. modiolus* spawning is known to be variable, both temporally and spatially. In Strangford Lough, Northern Ireland, slight spawning is known to occur year-round, with no apparent peak (Seed & Brown 1977; Brown 1984). Conversely, in Scandinavia, a spawning peak occurs in June, followed by a period of gonad redevelopment. Spawning is temperature dependent and is reported to occur within a narrow temperature range (7-10 °C). It is thought that the relatively constant temperatures in Strangford Lough facilitate the year-round spawning (Brown 1984). *M. modiolus* in the Irish Sea off the SE coast of the Isle of Man has been observed to follow an annual cycle of gonad development with a peak occurring in spring/summer, with trickle spawning occurring all year round (Jasim & Brand 1989).

IV.3.2. VULNERABILITY & THREATS

This section is divided up into (1) the vulnerability and (2) the threats (biological, chemical and physical) to each species in turn: *Sabellaria spinulosa; Sabellaria alveolata; Mytilus spp.* and *Modiolus modiolus.*

In this section, we refer to the sensitivity, vulnerability and potential for recovery of the habitat to sea level rise and storm events. In the case of natural reefs, flooding is not applicable and so is not discussed here. Much of the information from this section was sourced from the Marine Life Information Network website (www.marlin.ac.uk). We have adopted the terminology used by MarLIN with definitions below. In the following sections, we have identified the factors that are most likely to be associated with sea level rise and storm events for each species. The 'intolerance', 'sensitivity' and 'recoverability' of each species are presented in table format.

'Intolerance' is the suscptibility of a habitat, community or species (i.e. the components of a biotope) to damage, or death, from an external factor. Intolerance must be assessed relative to change in a specific factor.

'Recoverability' is the ability of a habitat, community, or species (i.e. the components of a biotope) to return to a state close to that which existed before the activity or event caused change.

'Sensitivity' is dependent on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery. For example, a very sensitive species or habitat is one that is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) and is expected to recover over a very long period of time, i.e. >10 or up to 25 years ('low'; recoverability). Intolerance and hence sensitivity must be assessed relative to change in a specific factor.

Sabellaria spinulosa

S. spinulosa is generally considered to be a very tolerant species with limited sensitivity (Table 4.1). Perhaps, the greatest sensitivity is to substratum loss, as once dislodged, the individual worms cannot rebuild their tubes. *S. spinulosa* is often one of the first species to recolonise an area after a disturbance (Cooper et al., 2007). Therefore, this species is expected to have a high recoverability.

S. spinulosa is most frequently found in polluted and disturbed conditions. *S. spinulosa* occurs in high densities on subtidal gravels that would be expected to be disturbed every year or perhaps once every few years due to storms and in polluted conditions. *S. spinulosa* appears to be very tolerant of water quality variation, but is potentially vulnerable to the short-term and localized effects of mineral extraction and the effects of oil dispersants on the larvae.

Factor	intolerance	Recoverability	Sensitivity	Confidence
Increase in temperature	Low	High	Low	Very low
Substratum loss	High	High	Moderate	High
Increase in suspended sediment	Low	Immediate	Not sensitive	Moderate
Increase in water flow rate	Intermediate	High	Low	Moderate
Increase in turbidity	Tolerant	Not relevant	Not sensitive	Low
Increase in wave exposure	Intermediate	High	Low	Moderate
Noise	Tolerant	Not relevant	Not sensitive	Low
Abrasion & physical disturbance	Intermediate	High	Low	Low
Displacement	High	High	Moderate	Low
Decrease in salinity	Intermediate	High	Low	Moderate
Changes in oxygenation	Intermediate	High	Low	Very low

Table 4.1. List of levels of 'intolerance', 'recoverability' and 'sensitivity' for physical and chemical threats to *S. spinulosa*. Level of confidence is included to give an indication of literature available on each factor. (Source: www.marlin.ac.uk. See Marlin website for primary sources)

Sabellaria alveolata

Similar to *S. spinulosa*, recolonisation of individuals *S. alveolata* is expected to be high, as long as there is suitable substratum for the settlement of larvae (Table 4.2). Recovery of reefs is expected to take considerably longer.

Factor	Intolerance	Recoverability	Sensitivity	Confidence
Increase in temperature	Intermediate	High	Low	Low
Substratum loss	High	Moderate	Moderate	Low
Increase in suspended sediment	Low	Very high	Very low	Low
Increase in water flow rate	Intermediate	High	Low	Low
Increase in turbidity	Tolerant	Not relevant	Not sensitive	Low
Increase in wave exposure	Intermediate	High	Low	Low
Noise	Tolerant	Not relevant	Not sensitive	High
Abrasion & physical disturbance	Intermediate	High	Low	Moderate
Displacement	High	Moderate	Moderate	High
Decrease in salinity	Intermediate	High	Low	Low
Changes in oxygenation	Intermediate	High	Low	Very low

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Mytilus spp.

Seed & Suchanek (1992) suggested that although mussel assemblages found in the upper intertidal or most sheltered sites, experience the least change per unit time, and may be considered more 'stable' (Lewis 1977), if disturbed, these assemblages would recover much slower than lower intertidal and more exposed sites. In addition, *Mytilus spp.* recovers quicker than other *Mytilus* species (Seed & Suchanek 1992). Overall, *Mytilus spp.* populations are considered to have a strong ability to recover from environmental disturbance (Table 4.3, Holt et al., 1998; Seed & Suchanek, 1992). Larval supply and settlement could potentially occur annually but settlement is sporadic with unpredictable pulses of recruitment (Lutz & Kennish 1992; Seed & Suchanek 1992). Therefore, while good annual recruitment is possible, recovery may take at least 5 years, although in certain circumstances and under some environmental conditions recovery may take significantly longer (Tyler-Walters 2008).

Modiolus modiolus

M. modiolus is a long-lived species and individuals are commonly observed to be older than 25 years. This species is regarded to be intolerant of loss of substratum, physical damage and abrasion (Table 4.4). Recovery is thought to take many years due to sporadic recruitment (Tyler-Walters 2001).

M. modiolus individuals or reefs are generally not thought to be fragile, however, physical threats from fishing gears pose a significant threat to this species. Older individuals are susceptible to boring by the sponge *Clione celata* which can make shells brittle, thus increasing vulnerability (Comely 1978).

Factor	Intolerance	Recoverability	Sensitivity	Confidence
Increase in temperature	Low	Very high	Very low	High
Substratum loss	High	High	Moderate	High
Increase in suspended sediment	Low	Immediate	Not sensitive	High
Decrease in emergence	Low	Very high	Very low	Low
Increase in water flow rate	Low	Very high	Very low	Moderate
Increase in turbidity	Tolerant	Not relevant	Not sensitive	Not relevant
Increase in wave exposure	Intermediate	High	Low	Moderate
Noise	Tolerant*	Not relevant	Not sensitive	Low
Abrasion & physical disturbance	Intermediate	High	Low	Moderate
Displacement	Intermediate	High	Low	Moderate
Decrease in salinity	Low	Very high	Very low	Moderate
Changes in oxygenation	Low	Very high	Very low	High

Table 4.3: List of levels of ‘intolerance’, ‘recoverability’ and ‘sensitivity’ for physical and chemical threats to *Mytilus spp.*. Level of confidence is included to give an indication of literature available on each factor. (Source: www.marlin.ac.uk. See Marlin website for primary sources).

Factor	Intolerance	Recoverability	Sensitivity	Confidence
Increase in temperature	Intermediate	Low	High	Very low
Substratum loss	High	Low	High	Moderate
Increase in suspended sediment	Low	Intermediate	Not sensitive	Low
Increase in water flow rate	Intermediate	Low	High	Low
Increase in turbidity	Low	Very high	Very Low	Moderate
Increase in wave exposure	Intermediate	Low	High	Very low
Noise	Tolerant	Not relevant	Not sensitive	High
Abrasion & physical disturbance	High	Low	High	Low
Displacement	Low	Very high	Very low	Very low
Decrease in salinity	High	Low	High	Moderate
Changes in oxygenation	Low	Very high	Very low	Moderate

Table 4.4. List of levels of ‘intolerance’, ‘recoverability’ and ‘sensitivity’ for physical and chemical threats to *M. modiolus*. Level of confidence is included to give an indication of literature available on each factor. (Source: www.marlin.ac.uk. See Marlin website for primary sources).

NATURAL AND ANTHROPOGENIC THREATS

These organisms are exposed to a broad range of threats; therefore not all are discussed in this section. Priority has been given to the most severe threats (physical, chemical and biological), with particular emphasis on those relating to floods and storms.

Physical threats can originate from natural and anthropogenic sources. Natural sources include increased temperatures, an increase in storm occurrence and intensity and sea-level rise, all of which are occurring as a result of global climate change. In this section we holistically address the general physical pressures faced by each species, rather than those from individual processes. Physical anthropogenic threats to reefs are extensive, not all are covered in this document. Some of the major threats to natural reefs are the impact of fishing gears, marine aggregate extraction, coastal development (including the construction of coastal defences), construction of offshore marine renewable and oil and gas exploration. Natural chemical threats posed by climate change include reduced salinity brought about by increased precipitation and surface runoff, acidification brought about by reduced pH and changes in oxygen concentrations. Anthropogenic chemical threats are primarily those associated with pollution. Biological threats are usually considered to be natural in the form of parasites, predators and competitors. However, invasion by non-native species is often as a result of human introduction and therefore can indirectly be considered an anthropogenic threat.

Sabellaria spinulosa

Physical threats

Sabellaria spinulosa usually occurs subtidally in areas of high water flow, and is relatively tolerant of wave and tidal-forcing. However, as *S. spinulosa* generally grows upon cobbles and pebbles (Connor et al., 2004), and it has been suggested that an increase in wave or tidal flow may reduce the stability of the attachment substratum resulting in increased scouring and mortality of individuals (Jackson & Hiscock 2008). It is a relatively disturbance-tolerant species and is often the first species to recolonise an area after a physical disturbance (Jackson & Hiscock 2008). The physical disturbance of removal from tubes and substratum loss will cause mortality. As *S. spinulosa* is predominantly subtidal, it is likely to be less affected by temperature changes than the intertidal *S. alveolata*, which has been shown to be severely affected by low winter temperatures (Crisp 1964). Fisheries for the pink shrimp *Pandalus montagui* and brown shrimps (*Crangon crangon*) (often associated with areas of *Sabellaria spinulosa* reefs) have been implicated in the loss or damage of reefs. However, Vorberg (2000) undertook experimental and observational studies that indicated only minor damage to tubes and rapid recovery as a result of shrimp fisheries. Nevertheless, populations, especially if as loose aggregations, may be displaced by mobile fishing gear.

Chemical threats

There is little data available on chemical threats to *S. spinulosa* although it is not thought to be sensitive to reduced salinity (Jackson & Hiscock 2008).

Biological threats

There is insufficient information available on biological threats to S. spinulosa.

Sabellaria alveolata

Physical threats

Sabellaria alveolata is typically found in the intertidal and is tolerant of changes in sediment regime. The physical disturbance of removal from tubes and substratum loss will cause mortality. Being an intertidal species, the greatest threats come from cold air temperatures and heavy wave action. It has been suggested that most colonies die through eventual break up by wave action (Jackson & Hiscock 2008). Increased exposure will result in potentially shorter colony life. *S. alveolata* is a southern species and is at the northern end of its range in Britain. This species is known to be negatively affected by extremely cold winters. In the cold winter of 1962/1963 *S. alveolata* suffered severe mortalities along the Welsh and southern English coastlines, where it had previously reached its northern and northeastern range limits (Crisp 1964). Populations suffered mortalities again during the winter of 1978/1979, but on a much smaller scale (Kendall & Bedford 1987). Recent work by Mieszkowska et al. (2006) showed that *S. alveolata* had re-colonised locations close to their northern range limits from where they were lost after the cold winter of 1962/1963. Despite the current trends in global warming, winter 2009/2010 was the coldest on record in Europe, which may have negatively affected *S. alveolata* at its range edges. Continued monitoring is necessary to detect future changes.

Chemical threats

There is insufficient information available on chemical threats to *S. alveolata*.

Biological threats

There is very little information available on the biological threats to *S. alveolata*. In a recent study of *S. alveolata* reefs in the Bay of Mont San-Michel, France found that reefs were becoming increasingly colonised by the invasive Pacific oyster *Crassostrea gigas* from local aquaculture operations and by green algae (*Ulva* spp.) due to the increasing inputs of nitrates from terrestrial origin (Dubois et al., 2006). It was found that epibionts, especially green algae, alter *S. alveolata* population structure, causing a reduction in new recruits that over the long run may cause significant damage to the reef structure itself. Furthermore, Dubois et al. (2006) noted that *C. gigas* have high filtration rates, suggesting that they may out-compete *S. alveolata* for food.

Competition for space with common mussels *Mytilus spp.* occurs, especially on boulder scars, but factors influencing this are unknown. Heavy settlement of mussels on *S. alveolata* reefs has been suspected of causing short term destabilisation and loss of habitat (Tyler –Walters 2008).

Mytilus spp.

Physical threats

Mytilus spp. can be found both intertidally and subtidally. It is a fairly tolerant species with the biggest threats posed by habitat loss and dislodgement by storms. Removal of the substratum, be it rock or sediment, will entail removal of the entire population and its associated community. Repeated substratum loss and recruitment results in a patchy distribution of mussels on the shore (Seed & Suchanek 1992). Storms and tidal surges are known to destroy mussel beds, often over hundreds of hectares in the Wash, Morecambe Bay and the Wadden Sea. With increasing wave exposure mussel beds become increasingly patchy and dynamic. *Mytilus spp.* beds may also be damaged by wave driven logs or equivalent debris (Seed & Suchanek 1992). Trampling by human traffic is most likely in spring and summer (Brosnan & Crumrine 1994). The combined effects of trampling and natural winter disturbances may result in loss of mussel beds in the long term. Displacement and or dislodgement by storms will likely lead to mortality. Dare (1976) found that individual mussels swept or displaced from rarely survived, since they either became buried in sand or mud, or were scattered and eaten by oystercatchers.

Chemical threats

In general, *Mytilus spp.* is tolerant of a wide range of contaminants and salinity and oxygen fluctuations. The most significant natural chemical threat to *Mytilus spp.* is a reduction in salinity caused by storm runoff (Hiscock pers. Comm. in Tyler-Walters 2008). The effects of contaminants on *Mytilus* sp. were extensively reviewed by Widdows & Donkin (1992) and Livingstone & Pipe (1992). Mussels are suspension feeders and, therefore, process large volumes of water together with suspended particulates and phytoplankton. Mussels absorb contaminants directly from the water, through their diet and via suspended particulate matter (Widdows & Donkin 1992), the exact pathway being dependant on the nature of the contaminant.

Biological threats

Mytilus spp. host a wide variety of disease organisms, parasites and commensals from many animal and plant groups including bacteria, blue green algae, green algae, protozoa, boring sponges, boring polychaetes, boring lichen, the intermediary life stages of several trematodes, copepods and decapods (Bower 1992; Gray et al., 1999). *Mytilus spp.* is threatened by a number of invasive species. *Aulocomya ater,* a mytilid, native to South America has been reported in the Moray Firth, Scotland in 1994 and again in 1997 (Holt et al., 1998; Eno et al., 2000; McKay 1994). *A. ater* is thought to have a stronger byssal attachment than *Mytilus spp.* and may replace *Mytilus spp.* in more exposed areas if it reproduces successfully (Holt et al., 1998).

The Pacific oyster *Crassostrea gigas* was introduced in Europe for commercial purposes in the mid 1960s. In Europe, wild populations of Pacific oysters are already found from northern Germany to southern Portugal. Fey et al. (2010) found that many mussel beds (*Mytilus spp.*) have been taken over by Pacific oysters in the Dutch Wadden Sea. In the German Wadden Sea almost all mussel beds are now considered oyster reefs (Nehls et al., 2006; Wehrmann et al., 2007). In the early stage of the development of *C. gigas*, Reise (1998) found 85% attached to *Mytilus spp.* (alive and empty shell) and 8% on other bivalves.

The American slipper limpet *Crepidula fornicata*, native to the North American East coast, was unintentionally introduced to Europe by oyster farming in the 1870s and now occurs from Denmark to Spain, Norway, the Mediterranean, Ireland and the United Kingdom (Blanchard 1997; Thieltges et al., 2003; Rayment 2007). There are conflicting results in the literature on the effects of *C. fornicata* on *Mytilus spp.*. In one set of field experiments (Thieltges 2005), the presence of *C. fornicata* has been shown to cause a reduction in survival and growth of the blue mussel, *Mytilus spp.*. A reduction in survival and growth of mussels was likely due to physical interference, associated with the attachment of *C. fornicata*. It is probable that when attachment onto a host occurs, the host organism will experience greater drag forces, requiring them to use more energy to remain attached to the substrate. This extra energetic requirement may result in reduced fecundity and survivability. Conversely, C. fornicata have also been found to benefit *Mytilus spp.*. Work done by the same authors, Thieltges (2005) found that *C. fornicata* presence on mussels led to a three-fold decrease in predation by starfish. Although starfish did not prey directly on *C. fornicata* it is believed that the cover provided by settled limpets made it more difficult for the starfish to prey on the mussels.

Modiolus modiolus

Physical threats

M. modiolus is thought to have an intermediate to high intolerance to physical disturbance (Tyler-Walters 2008 and 2007 respectively). Subtidal *M. modiolus* beds are susceptible to damage from fishing activities. In Strangford Lough, Northern Ireland, *M. modiolus* beds have been shown to suffer damage and mortality by scallop dredging (Service & Magorrian 1997; Magorrian & Service 1998).

Chemical threats

There is insufficient information available on chemical threats to *M. modiolus*.

Biological threats

Predation by crabs and starfish presents one of the greatest threats to juvenile *M. modiolus* (Brown & Seed 1977; Anwar et al., 1990; Tyler-Walters 2007). As mussels grow and become more difficult to open, the threat of predation becomes less important (Seed & Brown 1978). High densities of the brittle star, *Ophiothrix fragilis*, are considered to be capable of having a detrimental effect on *M. modiolus* beds not only through removal of both food and mussel larvae from the water column (George & Warwick 1985; Holt et al., 1998).

IV.3.3. KEY PROCESSES TO FOCUS ON FOR MAINTAINING ECOSYSTEMS INTEGRITY

In this section we discuss the processes to focus on for maintaining ecosystems integrity in terms of reefs in general and do not go into specifics for each species.

The spatial and temporal distribution of biogenic reefs can vary on vary small scales (i.e. metres and days) (Foster-Smith 2000; Foster-Smith & White 2001) making it difficult to accurately assess their status using point sampling methods. The ephemeral and unpredictable nature of biogenic reefs poses a challenge to effective management. The establishment of designated sites to protect habitats relies on a certain level of stability. Unless conservation effort can be concentrated on reefs of proven stability, site designation for biogenic reefs can prove unsuccessful.

Hendrick et al. (2011) suggest the designation of a much broader site comprising areas which already support dense populations or are considered suitable for potential biogenic reef development may be more beneficial. This approach is analogous to protection of mobile species rather than habitats or sessile species, affording protection of the environmental condition and mechanisms which enable biogenic reefs to develop. An alternative approach suggested by Hendrick et al. (2011) is the smaller-scale conservation of specific reef sites, with the view to the designation status lasting only for the lifetime of the actual reef. In order for this approach to be effective, the designation procedure must be on a shorter time scale (months rather than years).

Ideally, a combination of the above two approaches would prove to be the most effective. This would involve regular mapping of biogenic reefs within a larger supporting boundary. Exclusion zones around the reefs could be established and managed.

IV.3.4. CURRENT MANAGEMENT PRACTICES

Biodiversity is of intense interest for managers and policy-makers. As such, The United Nations declared 2010 the International Year of Biodiversity (Resolution 61/203). Throughout the course of the year events will take place world-wide to raise public awareness of not only the biological diversity on our planet, but the importance of protecting it. The origins of legal mechanisms and targets for protecting biodiversity mostly stem from the Convention on Biological Diversity (CBD) that was drawn up in 1992. Parallel to the CBD, the European Community (EC) adopted the Council Directive 92/43/EEC in 1992, this legalisation became more commonly known as the Habitats Directive. The directive focussed on the conservation of natural habitats and of wild fauna and flora through the establishment of a network of Special Areas of Conservation (SACs). The primary objective of which, is to promote the safeguarding and preservation of threatened species and habitats deemed to be of European importance.

In response to the CBD, the UK Government also founded the UK Biodiversity Partnership to develop and implement UK Biodiversity Action Plans (UK BAP). UK BAP recognises threatened biological assets within the UK and its surrounding waters and presents policies for the management and conservation of these assets. Plans for species and habitats in danger have been established to aid in recovery in order to assist in the UK's development in reducing biodiversity loss set out in the CBD (UK Biodiversity Group, 1999). To date it has lead to the construction of action plans for 1150 priority species and 65 priority habitats (BRIG, 2007). Reefs are one of the habitats listed under Annex I of the Habitats Directive which require the designation of an SAC.

Sabellaria spinulosa

Intertidal protection for *S. alveolata* reefs can be achieved through SSSI designation. *S. alveolata* reefs also occur as sub-features of non-reef Annex 1 habitats (eg intertidal mudflats and sandflats) under the Habitats Directive and are present in a number of candidate Special Areas of Conservation (cSACs). Discharges to the sea are controlled by a number of EC Directives, including the Dangerous Substances, Shellfish (Waters), Integrated Pollution Control, Urban Waste Water Treatment, and Bathing Waters Directives. The forthcoming Water Framework Directive will also be relevant. The Oslo and Paris Convention (OSPAR) and North Sea Conference declarations are also important. These commitments provide powers to regulate discharges to the sea and have set targets and quality standards to marine waters. An extensive set of standards covering many metals, pesticides and other toxic, persistent and bioaccumulative substances, and nutrients have been set under UK legislation.

Sabellaria alveolata

Intertidal protection for *S. alveolata* reefs can be achieved through SSSI designation. *S. alveolata* reefs also occur as sub-features of non-reef Annex 1 habitats (eg intertidal mudflats and sandflats) under the Habitats Directive and are present in a number of candidate Special Areas of Conservation (cSACs). Discharges to the sea are controlled by a number of EC Directives, including the Dangerous Substances, Shellfish (Waters), Integrated Pollution Control, Urban Waste Water Treatment, and Bathing Waters Directives. The forthcoming Water Framework Directive will also be relevant. The Oslo and Paris Convention (OSPAR) and North Sea Conference declarations are also important. These commitments provide powers to regulate discharges to the sea and have set targets and quality standards to marine waters. An extensive set of standards covering many metals, pesticides and other toxic, persistent and bioaccumulative substances, and nutrients have been set under UK legislation.

Mytilus spp.

Alhough *Mytilus spp.* is not designated under any protection laws, the habitat "Intertidal *Mytilus spp.* beds on mixed and sandy sediments" has been listed on the OSPAR List of Threatened &/or Declining Species and Habitats. *Mytilus spp.* is also protected by fisheries regulations. Fisheries regulations vary greatly in different parts of the Europe. The regulatory considerations in terms of mussel fisheries management are complex

Modiolus modiolus

In addition to its listing by OSPAR, this habitat is the subject of several local, national and regional listings, including the Habitats Directive (as part of 'Reefs') and the UK Biodiversity Action Plan. Such listings serve to highlight the conservation needs of the habitat, but successful protection depends on specific actions that follow. In the UK *M. modiolus* beds are identified as features for protection in SACs (Special Areas of Conservation) off Scotland, Wales and Northern Ireland.

V. OVERVIEW OF EXPERIMENTS (STATUS AND PRELIMINARY RESULTS) ON HOW TO IMPROVE CURRENT MANAGEMENT PRACTICES AND POSSIBILITIES TO RESTORE OR CREATE THESE ECOSYSTEMS

Within this section, we provide a concise overview of the ongoing and planned experimental work currently carried out within the THESEUS project, which should result in the insights needed to create the toolbox for coastal managers. These experiments combine a range of contrasting techniques, going from field to laboratory. We list the work per ecosystem.

V.1. DUNES

Concerning experimentations within the THESEUS project, no results can be presented yet. But, we would like to tackle the subject of dune vegetalisation with the intention of restoration or creation of dune habitat.

Coastal dunes are both valuable and vulnerable ecosystems. They have a strong patrimonial and landscape interest, and also play an important role en coastal protection. Dunes act as a barrier to storm surges and flooding. They also constitute a sandy stock for the beaches useful to face coastal erosion.

Submitted to strong natural and anthropic pressures, coastal dunes are damaged on almost all the European coasts. The restoration of degraded and disrupted coastal dunes is usually based on ecological engineering methods, using natural processes to improve the regeneration and the reinforcement of the coastal ecosystem. As dune vegetation is a key factor to coastal dunes stability, restoration projects aim to support native vegetation cover. This revegetation can be used on bare dunes that have been rebuilt or on degraded systems.

Experiments on vegetalisation by cuttings had been realized already (with *Ammophila arenaria* for instance). This method is besides acknowledged today and is widely used. We choose to experiment cuttings and seedbeds: seeds collected on sites near the experimental area, in order to conserve an ecological continuum of habitat. Direct seeding in comparison of seedlings plantings, makes sense in terms of efficiency and economics. This will also permit to considerably reduce the management cost of dune restoration or creation because the nursery expenses will be not to take into consideration anymore.

We will compare the germination and survival rates of different dunes species, in different conditions and with various methodologies (Table 5.1).

			Species			
Methods	Slope	Season	*Ammophila arenaria*	*Elymus farctus*	*Helichrysum stoechas*	*Artemisia campestris*
Cuttings	High slope	Automn	x	x		
		Spring	x	x		
	Low slope	Automn	x	x		
		Spring	x	x		
Cuttings + Geotextile	High slope	Automn	x	x		
		Spring	x	x		
	Low slope	Automn	x	x		
		Spring	x	x		
Seedings in holes	High slope	Automn				
		Spring				
	Low slope	Automn	x	x	x	x
		Spring	x	x	x	x
Seedings in holes+ geotextile	High slope	Automn	x	x		
		Spring	x	x		
	Low slope	Automn	x	x	x	x
		Spring	x	x	x	x
Scattering seeds	High slope	Automn				
		Spring				
	Low slope	Automn	x	x	x	x
		Spring	x	x	x	x
Scattering seeds + geotextile	High slope	Automn	x	x		
		Spring	x	x		
	Low slope	Automn	x	x	x	x
		Spring	x	x	x	x

Table 5.1. Methodology for dune revegetalisation experiments

V.2. SALT MARSHES

V.2.1. DEVELOPING INDICATORS FOR LONG-TERM STABILITY OF SALT MARSHES BY AERIAL IMAGES

With ongoing changes in climate and sea level rise, it is likely that weather and hydrodynamic patterns will change over time. For example, it is predicted, and already shown in empirical data (Young et al., 2011) that storminess increases. Due to this change, coastal ecosystems will be increasingly exposed to waves, which may have a large impact on the abundance and distribution of these valuable ecosystems (MEA, 2005). How salt marshes will respond to these changes is an ongoing debate, but there is a general consensus that salt marshes can keep up with sea-level rise if supplied with enough suspended sediments (Kirwan & Temmerman 2009). However, most salt marshes go through cycles of development and lateral erosion due to these natural sediment dynamics (van de Koppel et al. 2005). The onset of large scale lateral erosion of the marsh in combination with inhibition of salt marsh recovery is of concern to the long-term stability of salt marshes. Therefore, it is important to disentangle to what extent lateral marsh dynamics are part of natural cyclic behavior and to what extent changes in environmental conditions may cause irreversible losses.

A study to the long-term development of salt-marshes along the Scheldt was conducted by the NIOO-KNAW, to determine the most important processes affecting the scale at which cyclic behavior occurs. Aerial images over a period of more than 50 years were compared. This GIS/remote-sensing analysis, which was combined with some field experiments and a conceptual model, showed that the space needed for rejuvenation of the marsh strongly depends on the hydrodynamic regime and the ability of vegetation to re-establish on the mud flat in front of the eroding marsh. If marshes are more strongly exposed to waves they will go through longer (spatial) cycles compared to marshes that are sheltered. This is caused by the thresholds in vegetation establishment resulting in a hysteresis effect which is stronger in exposed salt marsh locations.

The manuscript is currently in an early stage of preparation, and data are not shown in this public document as this would hamper publication in a scientific journal. Preliminary results will be made available on request to the EU-commission (contact NIOO-KNAW: t.bouma@nioo.knaw.nl).

V.2.2 IDENTIFYING FACTORS LIMITING MARSH ESTABLISHMENT BY COMPARING 10 EUROPEAN SALT MARSHES

In order to gain a fundamental understanding of factors limiting salt marsh establishment and extension, we carry out a joined experiment to identify which factors that hamper establishment of pioneer vegetation. This means that we will deploy an identical experimental set-up at different locations across European sites (see fig. 5.1). We will measure the survival of rhizome segments, that will be planted along the elevation gradient (from mid marsh onto bare mudflat) at salt marshes with contrasting hydrodynamics exposure. The main focus is to correlate survival to measurements on the local sediment dynamics and their growth performance. Growth performance will be measured in a separate correlated experiment using a marsh organ (details described in section further down).

The techniques has been originally developed and tested in the Scheldt estuary. The manuscript describing the method and analyses technique is currently in preparation. Data are not shown in this public document, as this would hamper publication in a scientific journal. Preliminary draft of the manuscript is on request available to the EU-commission (contact NIOO-KNAW: t.bouma@nioo.knaw.nl). The additional measurements on the European scale offer an unique opportunity to test if the conceptual model that has been developed at a single site is applicable across Europe.

Country	**Estuary**	**location**	**Wave exposure**	**Species**	**Start exp.**	**End exp.**	**Responsible institute**
Netherlands	Scheldt	Baarland	Exposed	*Spartina anlgica*			NIOO-KNAW
		Zuidgors	Exposed	*Spartina anlgica*			NIOO-KNAW
		Paulina	Sheltered	*Spartina anlgica*			NIOO-KNAW
UK	Wales	Maltraeth	Sheltered	*Spartina anlgica*	May 2011	Apr 2012	BU
		Y Foryd	Sheltered	*Spartina anlgica*	June 2010	Apr 2012	BU
		Morfa Aber	Exposed	*Spartina anlgica*	May 2011	Apr 2012	BU
		Red Wharf Bay	Exposed	*Spartina anlgica*	June 2010	Apr 2012	BU
	Cornwall			*Spartina anlgica*			Plymouth
				Spartina anlgica			Plymouth
Spain				*Spartina alterniflora*			UC
				Spartina alterniflora			UC
Italy				*Spartina alterniflora*			UniBo

Table 5.1. List of sites and partners that will contribute to this experiment on seedling survival.

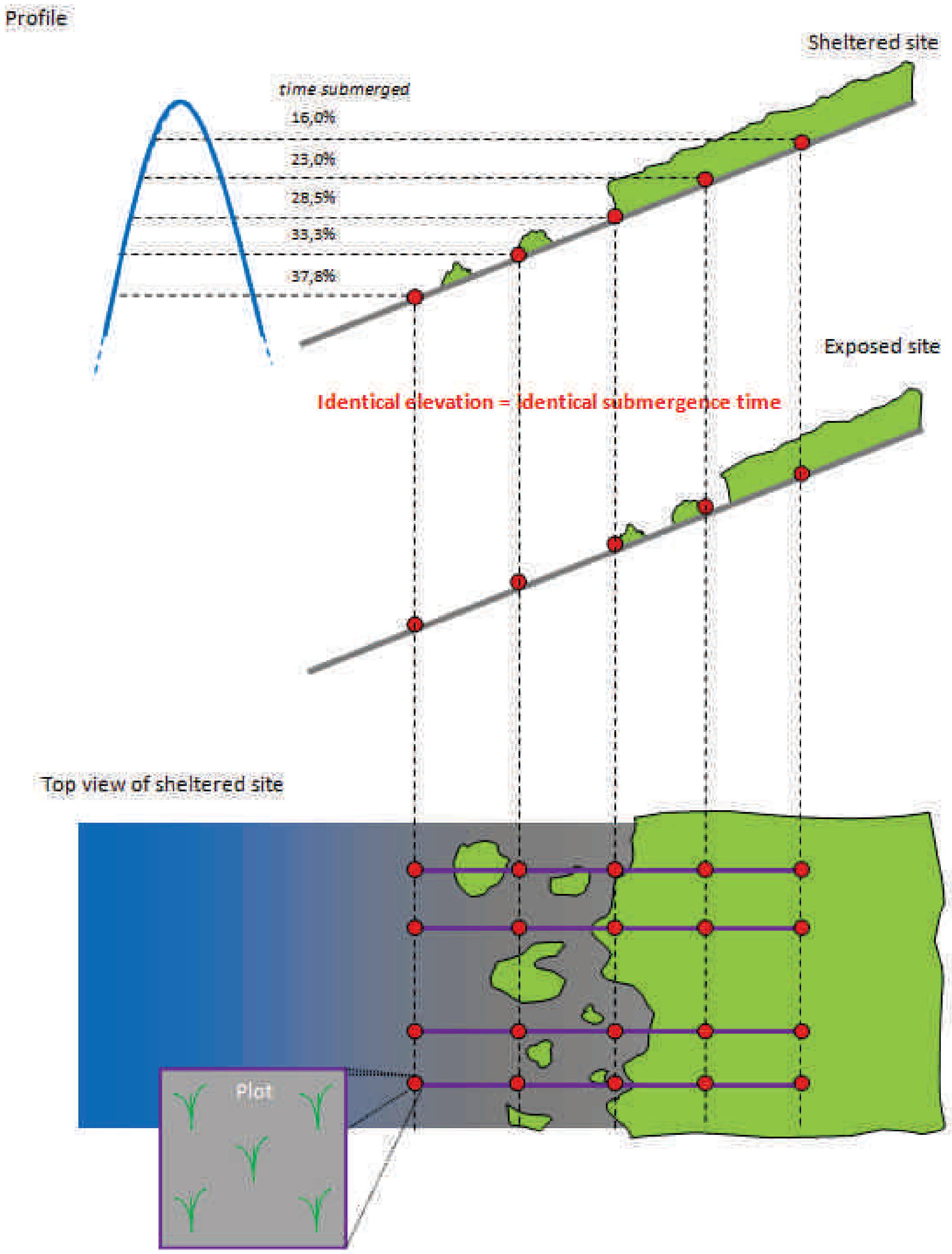


Figure 5.1. set-up of plots along elevation gradient of exposed and sheltered salt marsh sites. Inundation duration among plots should be the same along for transects.

V.2.3 DEVELOPING MECHANISTIC INSIGHT IN CRITICAL FACTORS TO SEEDLING ESTABLISHMENT: A FLUME STUDY

Coastal vegetation like salt marshes, mangroves, and sea grass beds, are valuable ecosystems for both their ecological function and their coastal protection capacity (Koch et al., 2009). These ecosystems are being threatened by increasing anthropogenic impact worldwide (Goodwin et al., 2001; Lotze et al., 2006; Orthy et al., 2006; Waycott et al., 2009). Moreover, sea-levels are rising, potentially causing drowning and dieback of many coastal wetland areas around the world (e.g. Nicholls, 2004). Only under right conditions, coastal wetlands are able to keep up with sea level rise through sediment accretion (French, 2006; Kirwan & Temmerman, 2009; Kirwan et al., 2010). Once coastal vegetations are lost, their restoration is usually very difficult (e.g. see Alongi 2002; Wolters et al., 2005; van Katwijk et al., 2009 and references therein). In-depth knowledge on essential factors determining (re-)establishment is required. However, little is known about the critical factors driving seedling establishment of sea-sea-grass beds and pioneer species in salt-marshes and mangroves

For both inter- and subtidal areas, it is well known that the sediment bed can show sedimentation and erosion at different time scale (e.g. see Smoak & Patchineeam, 1999; Yang et al., 2008). As seedlings are small and hence only are anchored in the top layer of the sediment bed, it may be expected that small disturbances like mixing of the sediment top-layer or short-term sheet-erosion events may cause seedling loss (Titus & Hoover, 1991). However, to our knowledge, until now very few studies have identified such sediment dynamics as a key factor in seedling establishment.

To be able to study the importance of sediment dynamics in seedling survival, NIOO-KNAW has developed an easy method to determine the critical sediment disturbance depth (CDD), defined as the minimum sediment erosion or mixing depth that restricts seedling survival. CDD is determined by subjecting seedlings to cumulatively increasing amounts of bed-erosion as a proxy for sheet erosion or sediment mixing and by exposing them to strong hydrodynamic forces to determine at which erosion depth plants fail. The concept was applied to salt-marsh pioneer *Spartina anglica*, mangrove pioneer *Avicennia alba*, and seagrass *Posidonia oceanic* and *Cymodocea nodosa*.

A manuscript describing this method is currently submitted for publication. Data are not shown in this public document, as this would hamper publication in a scientific journal. The submitted manuscript is on request available to the EU-commission (contact NIOO-KNAW: t.bouma@nioo.knaw.nl). This method provides important results to the THESEUS project as it will generate data that directly feed thresholds into the toolbox for managers.

V.2.4 RESILIENCE AGAINST DISTURBANCE: SEA-LEVEL RISE EXPERIMENT & MODELING AT 8 EUROPEAN MARSHES

With the ongoing change in climate and sea level it is likely currents and weather patterns will change which will affec marsh development and resilience against disturbance. Recent studies reveal that marshes inevitably go through cycles of lateral erosion and recovery. The recovery phase will be extremely critical to the development of the marsh because vegetation settlement can be hampered by changing environmental conditions. Together with the seedling establishment experiments across the 10 European sites the so called “marsh-organ” will generate data on the recovery potential of the marsh vegetation. Estimating the vitality of vegetation by using marsh-organs (see fig.5.2) is a proven method to study the effect of inundation on plant growth (Morris et al., 2002). Regarding the relative simple design, it will be possible to compare the growth of the same species across European sites. This would allow comparing the effect on seedlings from change in sea level (shore elevation) and the interaction with salinity (e.g. comparing locations in the Scheld estuary, Netherlands, tidal amplitude (comparing the UK with high tidal range, the Netherlands with intermediate tides, and Italy with micro tides) and climatic zones (NL & UK vs. France, Spain, & Italy).

Figure 5.2. The original “marsh-organ”, as designed by Jim Morris. This is not the same set-up that will be used by us.

Spartina development are monitored by taking photo’s every two weeks, and are send by mail to NIOO-KNAW contacts for comparison and decision on continuation of experiment. After a growing period of 2 to 4 months, dependent on growing progress monitored by photographs, the plants are collected from the pots and a number of biometrics per pot is measured quantitatively and qualitatively in the laboratory.

Country	Estuary	location	TIDAL AMPLITUDE	SPECIES	REPONSIBLE PARTNER
NL	Scheldt		4-6 m	Spartina anglica	NIOO-KNAW
	Wadden sea			Spartina anglica	NIOO-KNAW
UK	Wales	Y Foryd, Maltraeth	4.7 m 4.2 m	Spartina anglica	BU
	Cornwall		10 m	Spartina anglica	UoP
Spain	Santader		4 m	Spartina alterniflora	UC
	Cadiz bay		4 m	Spartina alterniflora	NIOO-KNAW
France	Gironde		5 m	Spartina alterniflora	EID
Italy	Adriatic sea		<1 m	Spartina alterniflora	UniBo
China	Yangtze			Spartina alterniflora	SKLEC

Table 5.2 List of eight sites that will participate in this experiment .

V.2.5 RESILIENCE AGAINST DISTURBANCE: RESISTANCE AGAINST WAVE ATTACK (NIOO-KNAW)

As described in section IV.2, recent insights in the development of salt-marshes indicate that the most important threats of sea-level rise to salt marshes may come from lateral erosion (Mariotti & Fagherazzi, 2010). That is, cliff formation in combination with coastal squeeze is one of the main risks for the long-term stability of salt-marshes. Aging of marshes will in general enhance the vulnerability of marshes to lateral erosion (van de Koppel et al., 2005). However, existing studies on cliff erosion mainly use conceptual modeling (e.g. Mariotti & Fagherazzi, 2010; van de Koppel et al., 2005; but see Callaghan et al. 2010). Hence, a mechanistic understanding of the process remains missing.

Based on the modeling work of Callaghan et al. (2010), we have developed a method to gain mechanistic insight in the way cliffs are being formed. Two experiments have been carried out during the winter in 2010, and the results are being analysed. A method description and preliminary results are on request available to the EU-commission (contact NIOO-KNAW: t.bouma@nioo.knaw.nl).

V.2.6 RESILIENCE AGAINST DISTURBANCE: EFFECT OF GRAZING MANAGEMENT (BU)

Plants provide the structural basis of the saltmarsh environment; they trap and root-bind the sediments (Kirwan & Temmerman 2008) and reduce wave energy (Möller et al. 1999). Wave attenuation improves the hydrological condition for further saltmarsh expansion, and sediment trapping may be crucial to keeping pace with sea level rise (Möller et al. 1999, Torres et al. 2006, Kirwan & Temmerman 2008). Root binding of sediments is likely to influence sediment stability, although there has been limited research to demonstrate this (Feagin et al. 2009). Clearly, the condition of the vegetation is of key imporance to the persistence of salt marshes (Kirwan & Temmerman 2008). This study will examine the effect of current grazing regimes on the state of the marsh vegetation. Resilience to grazer disturbance is evaluated with particular reference to vegetation attributes that are indicative of the ability of salt marshes to protext the coast (i.e., wave attenuation and induced sediment stability). It is assumed that diminished attributes for coastal protection represent a risk for is likely to drive towards a condition where marshes will be at risk from erosion. Thus, the study will focus on examining the effects that grazing has on vegetation attributes that are of specific importance to coastal protection.

It is suspected that management of grazing regime has the potential for influencing saltmarsh persistence, because grazers are powerful regulators of vegetation state (Bakker et al. 2005, Milotic et al. 2010). Grazer influence vegetation growth and biomass directly via feeding and soil disturbance (Bakker, et al., 2005; Kiehl, et al., 2007) and indirectly via trampling, which compacts the soil and reduces oxygen content (Vera, 2000). Grazing impact on composition and structure of vegetation varies with stocking rate, timing of grazing and grazer composition (Milotic et al. 2010). Vegetation patchiness and biodiversity peaks at light grazing intensity (Bakker et al. 1985, Kiehl et al. 1996). Intense grazing transforms complex communities with woody species into homogenous lawns dominated by short flexible grass (Andresen et al. 1990, Kiehl et al 1996). Flume work and single species tests suggest that wave attenuation is diminished in such communities (Bouma et al. 2005, Möller 2006). Light grazing increases vegetation patchiness (Bakker et al.1985, Kiehl et al 1996), which might cause local turbulence thereby reducing local sedimentation (Boorman et al. 1998, van Wesenbeeck et al. 2008). The effect of patchiness on wave attenuation is not known. One study suggested intense grazing diminished sediment accretion (Neuhaus et al. 1999). Conversely, grazing can stimulate root growth (Frank et al. 2002), thus boosting below-ground contributions to marsh surface elevation and sediment binding, and livestock trampling compacts the soil which might enhance cohesion and reduce erodability (Meyles et al. 2006).

We hypothesise that low grazing intensity will be optimal for coastal protection beause it: provides sufficient above and below ground vegetation cover to attenuate waves and bind sediments; it will reduce surface erodability via compacting the soil. We will test this hypotheses by sampling a series of soil attributes of relevance to wave attenuation and surface erosion: above-ground vegetation height and cover (proxy of wave attenuation), below ground root density and depth (proxy of soil binding), soil physical state (granulometry and compaction) and soil oxic state (proxy of vegetation growth potential). For a sub-set of marshes per level of grazing intensity, we will deploy simple instruments (Gypsum Clod Plates and Denny Dynamometers, see below) for direct measures of wave energy accross the marsh.

Some of the study marshes will be included in the 2.1, 2.2, and 2.4 experiments, which will provide breadth of resolution for the assessment of impact on grazing on marsh functioning. For instance, the instrumented measurements of wave attenuation across the marsh edge (see III.2.2) will include five of the 23 marshes sampled here.

Aims and objectives. A broad-scale study of up to 23 regional marshes will examine the effect of grazing intensity on proxy-parameters of coastal protection by marshes. The study will identify the grazing management regimes that are most likely to retain the marsh ecosystem service of coastal protection. Specific objectives are: (i) To quantify how different grazing regimes control vegetation and soil attributes in regional marshes; (ii) To examine the effect that these attributes might have on parameters of coastal protection by marshes. *Study region.* This work will be done by Bangor University. It will focus on marshes in Wales and NW England, as this region has predominance of marshes that are exposed to grazing. The Geographical area from Dyfi Estuary to Morecambe Bay in the Irish Sea (fig 5.2) has well characterised inshore hydrodynamic and water quality regimes. Many marshes have been agriculturally grazed for generations; others are without significant domestic grazing. Some are managed with different grazing regimes as part of agri-environmental interventions (e.g. Tir Gofal in Wales) or for planned conservation outcomes (e.g. bird reserves of the Royal Society for the Protection of Birds). The region lies within a single biogeographically defined zone for saltmarsh vegetation. Thus, there are a range of biogeographically uniform marshes with similar geomorphological settings and hydrodynamic and sedimentological regimes, but where grazing differs. Our studies will exclude marshes where salinity is <15 and marshes fringing the open coast (regionally rare). *Grazing regimes.* We focus on marshes grazed by sheep (regionally predominant). 'Grazing regime' will be categorised into four levels, based on the categories used by the organisations responsible for licensing agreements (Natural England, NE; Countryside Council for Wales, CCW): Ungrazed (UG), Lightly Grazed (LG: max 3 sheep ha^{-1} y^{-1}; no grazing in March-July), Moderately Grazed (MG: max 7 sheep ha^{-1} y^{-1}, reduced to 3 ha^{-1} y^{-1} in March-July) and Intensively Grazed (IG: recommended max of 13 sheep ha^{-1} y^{-1}; no time restrictions).

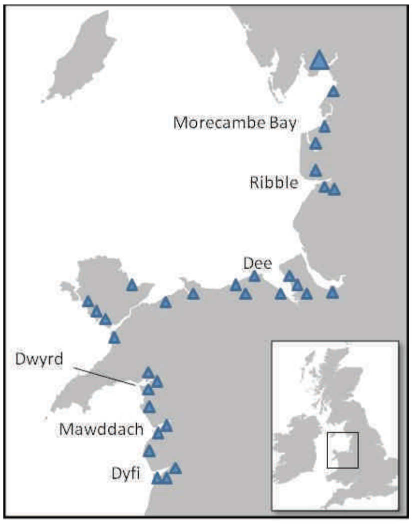


Figure 8.2. Study sites in Wales and West England for examining the influence of grazing management regimes on the proficiency of coastal protection by salt marshes.

Sampling design. This survey will incorporate up to 23 marshes that fit our criteria. We will collate existing data on hydrology (incl. wave energy exposure). Data on grazing regimes and grazing history will be derived from land owners and license organisations. We will sample locations at the mid and high marsh (where present). These levels are the main grazing zones; the low marsh is not consistently grazed. *The following will be sampled*. Vegetation composition and structure. Plant species, cover and height will be recorded in ten random 4 m^2 quadrats/zone. Soil compaction will be measured with a penetrometer and soil stability by a shear-vane tester in 4 of the 10 quadrats/zone. Root biomass will be sampled in these four quadrats using a 50 mm core, inserted to 50 cm in the cropped area. We will measure redox potential (indication of soil oxygen content) at 5, 10, 15, 25 and 50 cm on core extraction. The core will be analysed in the lab for: sediment bulk density, salinity, soil moisture and granulometry; the depth, density and diameter of roots will be recorded with a soil scanner. We will use Gypsum Clod Plates and Denny Dynamometers (DD) to provide relative measures of average and maximum wave-energy. Clods have been used for this purpose in marshes (Bouma et al. 2008) and DDs have been used on rocky shores (Denny 1983). They will be deployed monthly for 12 months, placing instruments immediately below the marsh edge and in the middle of the low, the mid and the high marsh. Change across the marsh in clod dissolution and max wave-energy (DDs) will provide indicative measures of attenuation of wave and water energy.

Analysis. We will test the main hypothesis using non-parametric multivariate analysis of variance (PERMANOVA; Anderson et al. 2008). We will look for effects of the four categorical levels of grazing regime on the ordination of marshes, as well as of individual sample variables. Patterns, such as vegetation community assemblages, will be explored by multi-dimensional scaling (MDS) and associated tools. The analysis will assume: (i) marshes with greater vegetation cover, below-ground biomass, soil compaction and rootnet density provide greater sediment stability (based on Feagin et al. 2009); and (ii) wave attenuation is positively correlated with above-ground biomass/vegetation height (based on Möller et al. 1999; Möller & Spencer 2002). Assumption (ii) will be tested directly using data from field observations of wave attenuation in section III 2.2 (which involves several of the study marshes), as well as Clod and DD observations.

Timelines and data analysis. Sampling was commenced in June 2010. The vegetation survey and sampling of soil physical parameters is complete. Root cores and granulometry samples are currently being processed. Gypsum clods and DD sampling will be completed by March 2012. Data analysis, excepting for clods and DDs, will be completed in autumn 2011.

V.2.7 THE APPLICATION OF CUTTING PLUS WATER LOGGING TO CONTROL SPARTINA ALTERNIFLORA ON SALT MARSHES IN THE YANGTZE ESTUARY

Control and eradication of the exotic and invasive plant Spartina alterniflora within the Chongming Dongtan nature reserve is vital for the management and conservation of the saltmarshes. A demonstration project was established at the nature reserve where controlled waterlogging and cutting were carried out to investigate effective measures for controlling this invasive plant on saltmarshes. The results during the years 2007-2008 showed that although the managed waterlogging itself could significantly reduce biomass and seed production of *S. alterniflora* at an early stage, this species showed a fast adaptation to the long-term waterlogging stress. Thus, managed waterlogging alone could not control or eradicate *S. alterniflora* effectively. On the other hand, managed waterlogging combined with cutting the aboveground part of *S. alterniflora* at a key stage (the florescence period in July) could effectively control and eradicate this plant. Both the aboveground and belowground parts of *S. alterniflora* were killed and the plants began to decompose after 3 months and there was no re-growth of the emergent part of *S. alterniflora* in the following years. Therefore, an integrated technique of cutting at the florescence period combined with managed waterlogging for about 3 months has been shown to be a realistic and effective technique for controlling and managing the invasion of *S. alterniflora* in the nature reserve. However, once the cofferdam that impounded the water for waterlogging was removed to restore the natural hydrodynamic regime of wetlands, the seeds and seedlings of *S. alterniflora* could reinvade the controlled site from the neighboring area and reestablish a new *S. alterniflora* community. Therefore, after eradication of *S. alterniflora*, proper measures should be taken to prevent its reestablishment. The results of this demonstration project provide a useful and effective approach to controlling and managing the large scale invasion of *S. alterniflora* on saltmarshes in the Yangtze Estuary, China.

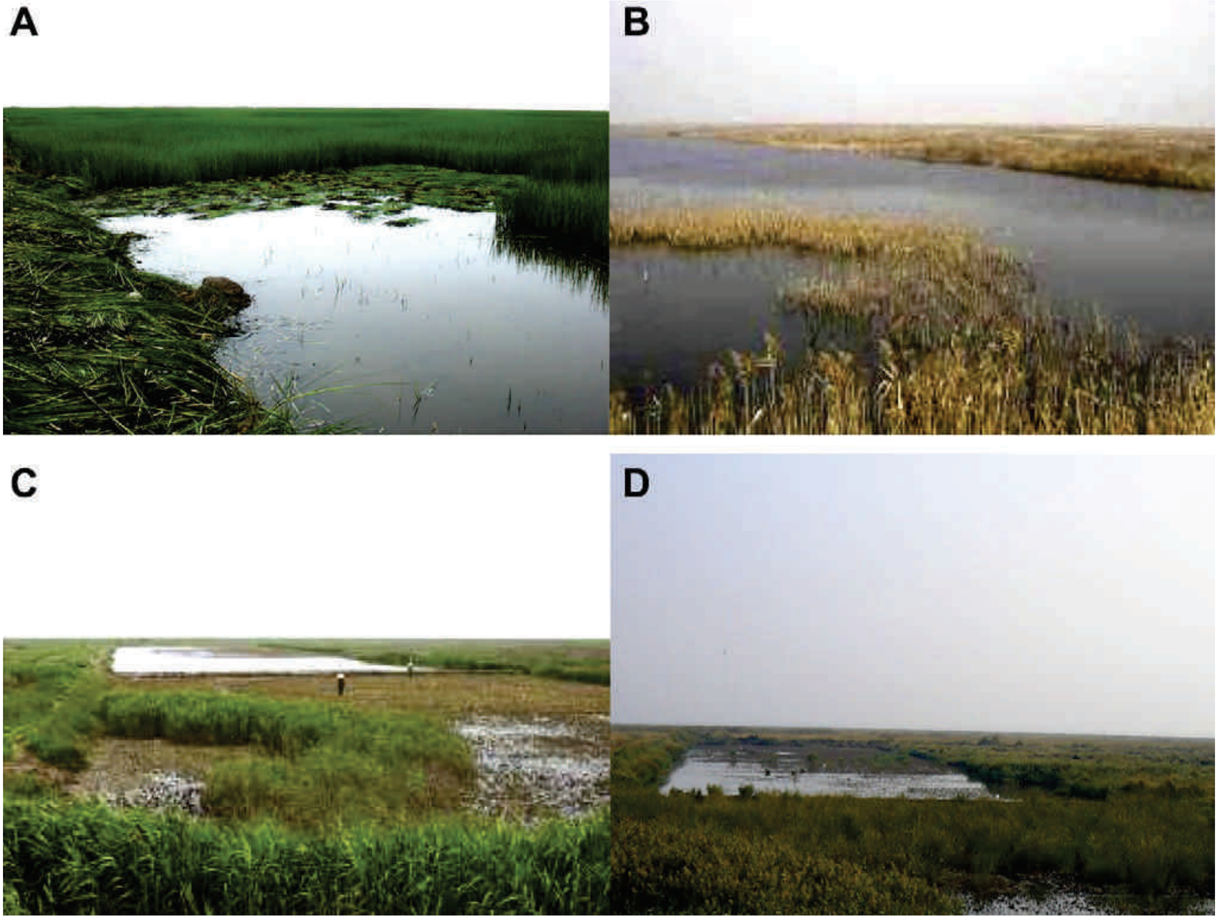


Figure 5.3. The effects of different treatments on the growth of S. alterniflora during the years 2007-2008 at the demonstration project site. A in July 2007, B in November 2007, C in June 2008 and D in October, 2008.

V.3. BIOGENIC REEFS

Our research on biogenic reefs are centred around sabellid worms, with the primary focus being *S. alveolata.*

Biogenic structures create complexity, increasing frictional stress that leads to reductions in flow speed and the development of turbulent conditions. There is no doubt that foundation species that form structures facilitate other species, and examples are found within many marine habitats such as tropical coral reefs, salt marshes, macroalgae, sea grasses and mussel beds. Some processes have received significant research effort, such as elevated local biodiversity in a range of structural habitats. It is often overlooked that elevated biodiversity is a result of more fundamental processes such as the provision of space and the environmental modification by the structure forming organism themselves. Individual- and population-level traits such as density and elevation drive the intensity of the interaction with flow. This has implications for other processes such as particle entrainment, dissolved nutrient replenishment and residence times of waste or larvae. Gaining a better understanding of how structures formed by *Sabellaria* spp. influence local hydrodynamics are a priority. Our general lack of knowledge of hydrodynamic interactions in *Sabellaria* spp. reef habitats is of great concern, especially in light of continued anthropogenic pressure.

Reef formation, evolution and decay

Structures created by *Sabellaria alveolata* consist of unbranched dwelling tubes that are constructed by individuals from suspended matter, usually sand grains and broken shell. In areas where *S. alveolata* form aggregations their tubes become intertwined to form rigid structures that can have different morphologies. The most common forms consisting of either horizontal growth that resemble crusts or vertical growth that stands proud of the seafloor to a maximum reported height of 50 cm. Many biogenic structures that protrude above the seafloor interact with near-bed flow, commonly resulting in particle trapping, but sometimes can also influence net erosion. This ultimately depends on several different factors, including local physical conditions such as flow velocity and particle behaviour, and is also affected by the size, shape and density of benthic structures. The ability to form a biogenic concretion and the resulting ecosystem function associated with particle trapping (including elevated local biodiversity) is becoming used as one of the defining characteristics of *S. alveolata* reefs; yet no

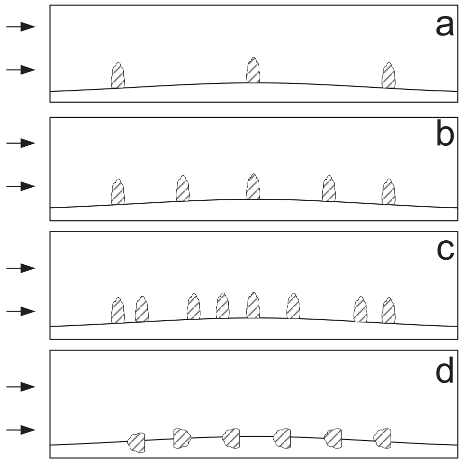


Figure 5.4. example experimental treatments for determining the sediment dynamic characteristics over *S. alveolata* structures; a, low density ordered aggregation; b, high density ordered aggregation; c, high density aggregation with randomised placement; d, broken aggregations simulating bottom impact.

experimental work has been published establishing to what extent *S. alveolata* structures alter near-bed conditions.

To address this gap in knowledge we will use the largest flume at Bangor University to create a series of simulated *S. alveolata* aggregations consisting of both living and dead structures (fig.5.4). These will be placed at varying densities, in both ordered and random placements and statistically controlled through the use of artificial structures with known surface complexities. Each experimental treatment will be maintained in the flume with several trials mimicking local environmental conditions including flow velocity and suspended sediment levels. This will allow, monitoring of sediment erosion, deposition and composition between the structures and recording of the hydrodynamic characteristics of the structures using arrays of UDVPs, to establish a baseline of how *S. alveolata* structures influence the local physical habitat. We will then elevate suspended sediment levels to simulate the heavy deposition associated with aggregate extraction, recording how this affects the dynamics of the reef structures, particle transport and burial.

In addition to the experimentally manipulated experiments, we will monitor where possible, how growth of *S. alveolata* structures in the flume progressively modifies the hydrodynamic characteristics, comparing how initial physical conditions are altered as reef elevation and complexity increases through structural growth.

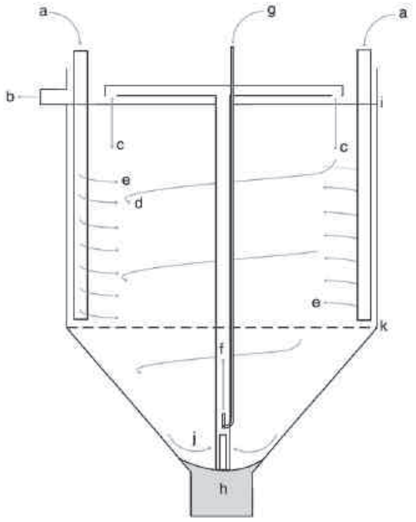


Figure 5.5. Schematic diagrams of the VoRT. Key to diagram: a. water inflow, b. water outflow, c. sediment outflow from airlift, d. Distribution of sediment by generated current, e. current generation, f. air or water lift, g. air hose or water hose, h. sediment sump, i. water level, j. sediment suction into lift, k. perforated mesh base for samples.

Environmental tolerances

Most organisms live within clearly defined environmental niches. When conditions extend outside of this niche; it becomes intolerable and can have an impact on their physiology, cumulating in disruption to metabolic processes and ultimately behaviour. Currently, the environmental tolerance of *S. alveolata* is unclear, specifically to sediment preference, temperature and water velocity. Without this baseline data, it is difficult to establish the true impact of anthropogenic activities from natural variability.

To address this uncertainty, we propose to use a series of compact, temperature controlled flumes known as vortex resuspension tanks (VoRTs, see fig.5.5) to subject colonies of living *S. alveolata* to a multifactorial experiment in order to assess the impact of varying flow velocity, sediment availability and temperatures on tube building capability and behaviour of *S. alveolata*. Samples will be maintained in the flumes under a set of flow velocities and temperatures. Samples will then be buoyant weighed to assess changes in mass every two days during the experimental period and, if possible, respiration rates will also be recorded during this time.

VI. TRANSLATING EXPERIMENTS TO APPLICATION: A TOOLBOX OF MANAGEMENT STRATEGIES ON DUNES, SALT MARSHES & BIOGENIC REEFS

Although some of the THESEUS deliverables are still a long time before being due, we realize it is important to early on communicate the type of output (units) that will be supplied, to facilitate exchange with other WPs. Hence, in this section, we try to summarize the anticipated outputs of WT 3.1 and 3.2 in a table.

Chapter. Paragraph	**ecosystem**	**Action, experiment, measurment**	**Toolbox application**	**Toolbox parameters**	**Progress**	**Available manuscripts**
III.1.2	dunes	Global assessment method	Coastal protection : Dune integrity evaluation	Patrimonial interest	done	Grosset, 2007
III.1.2	dunes	Global assessment method	Coastal protection : Dune integrity evaluation	Conservation status	done	Grosset, 2007
III.1.2	dunes	Global assessment method on dune revegetalisation	Coastal protection : Dune integrity evaluation	Comparison of indicators before and after vegetalisation	In progress	
III.2.2	marsh	Wave attenuation over salt-marsh, Yangtze delta, China	Coastal defense value at high waves environments	Wave height decay function	done	Yang et al. submitted
III.2.2	marsh	Wave attenuation over salt-marsh, Yangtze delta, China	Coastal defense value of stiff vs. flexible vegetation	Wave height decay function	done	Ysebaert et al., submitted

III.2.2	marsh	Wave attenuation over salt-marsh, Wales, UK	Coastal defense value of, and resilience of salt-marshes with different mudflat-marsh landforms	Wave height decay function	in progress (some measurements delayed due to technical issues)	-
III.3.2	reefs	Wave attenuation over oyster reef, Eastern Scheldt, the Netherlands	Coastal defense value of oyster reefs	Wave height decay function	pilot done, data need to be analyzed (jan/feb 2010)	
III.3.2	reefs	Wave attenuation over oysterreef, Wadden sea, the Netherlands	Coastal defense value of oyster reefs	Wave height decay function	Scheduled for autumn/winter 2011	
III.3.2	reefs	Wave attenuation over mussel beds, Wadden sea, the Netherlands	Coastal defense value of mussel beds	Wave height decay function	Scheduled for autumn/winter 2011	
III.3.2	reefs	Wave attenuation over sea grass, flume experiment with WP 2	Coastal defense value of sea grass	Wave height decay function	Scheduled /done???	
III.3.2	reefs	Wave attenuation over *Sabellaria*, North Wales	Coastal defense value of *Sabellaria*	Wave height decay function	Scheduled autumn/winter 2011	
V.1.	dunes	Dune restoration and resilience to sea	Coastal defense: Experiments on vegetalisation by	Runup decay function	In progress	

		flooding	cuttings			
V.1.	dunes	Dune restoration and resilience to sea flooding	Coastal defense: Experiments on vegetalisation by seeding	Runup decay function	In progress	
V.1.	dunes	Dune restoration and resilience to sea flooding	Coastal defense: Experiments on vegetalisation by scattering	Runup decay function	In progress	
V.2.1	marsh	Developing indicators for long-term stability of salt-marshes by aerial images	Spatial indicators for assessment of salt-marsh vegetation 'health'		done	van Belzen et al., in prep
V.2.2	marsh	Identifying factors limiting marsh establishment by comparing 10 European salt-marshes	Identification of stressors (stresses affecting growth rate/primary production, and disturbances)	critical sediment disturbance depth x innundation level	in progress	In Scheldt estuary: van Belzen et al., in prep.
V.2.3	marsh	Developing mechanistic insight in critical factors to sedling estabilshement: a flume study	Deriving critical sediment disturbance depth parameters for vegetation establishment and development	critical sediment disturbance depth	done	Bouma et al., submitted.
V.2.4	marsh	Resilience against disturbance: sea-level rise experiment & modeling at 8 European marshes	Vulnerability assessment for climate change and sea level rise at different scenarios		in progress	

V.2.5	marsh	Resilience against disturbance: resistance against wave attack	Determine mechanism that result in the onset of cliff erosion	critical sediment disturbance depth	in progress	
V.2.6	marsh	Resilience against disturbance: effect of grazing management	Determine levels of sustainable grazing/stocking densities		?	
V.2.7	marsh	The application of cutting & water logging to control Spartina alterniflora	Knowledge on how to prevent invasive Spartina species from colonizing coastal areas		in progress	
V. 3	reefs	Resilience against disturbance: fragmentation effect on hydrodynamics of *Sabellaria*	Vulnerability assessment for climate change and sea level rise at different scenarios	Reef quality versus erosion	Summer-winter 2011	
V.3	reefs	Resilience against disturbance: Establishing sediment , temperature and flow preference for *Sabellaria*	Vulnerability assessment for climate change and sea level rise at different scenarios	Sediment preference profiles. Temperature range. Flow preference.	In progress	

VII. LITERATURE

AAGARD T., ORFORD J., and MURRAY A.S., 2007. Environmental controls on coastal dune formation; Skallingen Spit, Denmark. Geomorphology, 83, 29-47.

ADAM P., 2002. Salt marshes in a time of change. Environmental Conservation, 29: 39-61.

ADNIT C, and 8 others, 2007. Saltmarsh management manual. Environment Agency, Bristol. ISBN: 978-1-84432-714-0 Downloadable from: www.defra.gov.uk/environ/fcd/research

ALLEN J.R.L., 2000. Morphodynamics of Holocene salt marshes: a review sketch from the Atlantic and Southern North Sea coasts of Europe. Quaternary Science Reviews, 19: 1155-1231.

ANDERSON M. J., GORLEY R. N., CLARKE K. R. 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E, Plymouth, UK.

ANDRESEN H., BAKKER J.P., BRONGERS M., HEYDEMANN B., IRMLER U., 1990. Long-term changes of salt-marsh communities by cattle grazing. Vegetatio, 89, 137–148

ANWAR N. A., RICHARDSON C.A., & SEED R., 1990. Age determination, growth rate and population structure of the horse mussel *Modiolus modiolus*. Journal of the Marine Biological Association of the United Kingdom 70, 441-457.

ARAUJO R., HONORADO J., GRANJA H. M., NEVES DE PINTO S., BARRETO CALDA F., 2002. Vegetation complexes of coastal sand dunes as an evaluation instrument of geomorphologic changes in the coastline. Littoral 2002, The changing Coast : 337-339. EUROCOAST/EUCC, Porto-Portugal : 337-339.

BAKKER J. P., DIJKSTRA M., RUSSCHEN P. T., 1985. Dispersa, germination and early establishment of halophytes and glycophytes on a grazed and abandoned salt-marsh gradient. New Phytologist 101, 291-308

BAMBER R.N. & IRVING P.W., 1997. The differential growth of *Sabellaria alveolata* (L.) reefs at a power station outfall, Polychaete research, 17, 9-14.

BERGHAHN R., VORBERG R., 1993. Effects of the shrimp fisheries in the Wadden Sea. In: Influence of fisheries upon Marine Ecosystems. Einfluss Der Fischerei Auf Marine Oekosysteme Lukowicz, M., 103-126.

BERTNESS M.D., ELLISON A.M., 1987. Determination of pattern in a New England salt marsh plant community. Ecol Monogr 57:129-14

BEUKEMA J.J., 1983. Quantitative data on the benthos of the Wadden Sea proper. In: Wolff, W. J. (Ed.). *Ecology of the Wadden Sea* 1. Balkema, Rotterdam: pp. 134-142. Original reference not seen. Cited by Flemming, B. & Delafontaine, M. (1994). Biodeposition in a juvenile mussel bed of the east Frisian Wadden Sea (Southern North Sea). Aquatic Ecology, 28: 289-297.

BEUKEMA J.J., CADEE G.C., 1996. Consequences of the sudden removal of nearly all mussels and cockles from the Dutch Wadden Sea. PSZN Mar Ecol 17:279–289

BLANCHARD M., 1997. Spread of the slipper-limpet (*Crepidula fornicata*) in Europe. Current state and consequences. Scientia Marina 61 (2 sup.): 109-118

BOORMAN L. A., GARBUTT A., BARRAT D., 1998. The role of vegetation in determining patterns of the accretion of salt marsh sediment. 1998. In: BLACK K.S., PATERSON D.M. , CRAMP A. (eds) Sedimentary Processes in the Intertidal Zone. Geological Society, London, Special Publications 139:389.

BOUMA T.J., DE VRIES M. B., LOW E., PERALTA G., TNCZOS I. C., VAN DE KOPPEL J., HERMAN P. M. J., 2005. Trade-offs Related to Ecosystem Engineering: A Case Study on Stiffness of Emerging Macrophytes. Ecology 86, 2187-2199

BOWER S.M., 1992. Diseases and parasites of mussels. In: Gosling, E. (Ed.). *The mussel Mytilus: ecology, physiology, genetics and culture*. Developments in Aquaculture and Fisheries Science 25. Elsevier, Amsterdam: pp. 543-563.

BODERE J.C., CRIBB R., CURR R., DAVIES P., HALLEGOUET B., MEUR C., PIRIOU N., WILLIAMS A., YONI C., 1991. La gestion des milieux dunaires littoraux. Evaluation de leur vulnérabilité à partir d'une liste de contrôle. Etude de cas dans le sud du Pays de Galles et en Bretagne occidentale. Norois n°151. Poitiers-France. 279-298

BRAITHWAITE C.J.R., ROBINSON R.J., & JONES G., 2006. Sabellarids: a hidden danger or an aid to subsea pipelines? Quarterly Journal of Engineering Geology and Hydrogeology, 39: 259-265.

BREWER J.S., GRACE J.B., 1990. Plant community structure in an oligohaline tidal marsh. Vegetatio 90: 93–107

BROSNAN D.M., & CRUMRINE L.L., 1994. Effects of human trampling on marine rocky shore communities. Journal of Experimental Marine Biology and Ecology, 177: 79-97.

BROWN R.A. 1984. Geographical variations in the reproduction of the horse mussel, *Modiolus modiolus* (Mollusca: bivalvia). Journal of the Marine Biological Association of the United Kingdom, 64, 751-770.

BROWN R.A. & SEED R., 1977. *Modiolus modiolus* (L.) - an autecological study. In: KEEGAN B.F., O'CEIDIGH P., BOADEN P.J.S. (eds). Biology of Benthic Organisms. Proceedings of the 11th European Symposium on Marine Biology, Pergamon Press, Oxford, Galway, Ireland, pp 93-100.

CALLAWAY R., ENGELHARD G. H., DANN J, COTTER J., & RUMHOR H., 2007. A century of North Sea epibenthos and trawling comparisons between 1902-1912, 1982-1895 and 2000. Marine Ecology Progress Series, 346: 27-43.

CARTER R.W.G., 1988. Coastal environments. An introduction to the physical, ecological and cultural systems of the coastlines. Academic Press, New York.617pp.

CASPERS H., 1950. Die Lebensgemeinschaft der Helgolander Austernbank. Helgoland Marine Research, 3: 119-169.

CHISHOLM J.R.M., & KELLEY R., 2001. Worms start the reef-building process. Nature, 409: 152-153.

CLEMMENSEN L. B., PYE K., MURRAY A., and HEINEMEIER J., 2001. Sedimentology, stratigraphy, and landscape evolution of a Holocene coastal dune system, Lodbjerg, NW Jutland, Denmark. Sedimentology, 48, 3-27.

COLEMAN N., 1976. Aerial respiration of *Modiolus modiolus*. Comparative Biochemistry and Physiology Part A: Physiology, 54: 401-406.

COMELY C.A. 1978. Modiolus modiolus (L.) from the Scottish west coast. Ophelia, 17, 167-193.

COMMITO J.A., CELANO E.A., CELICO H.J., COMO S., & JOHNSON C. P., 2005. Mussels matter: postlarval dispersal dynamics altered by a spatially complex ecosystem engineer. Journal of Experimental Marine Biology and Ecology 316(2): 133-147.

COMITO J.A., RUSIGNUOLO B.R., 2000. Structural complexity in mussel beds: the fractal geometry of surface topography. Journal of Experimental Marine Biology and Ecology 255: 133–152.

COMMITO J.A., DANKERS N., 2001. Dynamics of spatial and temporal complexity in European and North American soft-bottom mussel beds. In: K. Reise, Editor, *Ecological Comparisons of Sedimentary Shores*, Springer-Verlag, Heidelberg, pp. 39–59.

CONNOR D.W., ALLEN J.H., GOLDING N., HOWELL K.L. LIEBERKNECHT L.M., NORTHEN K.O. & REKER J.B., 2004. The Marine Habitat Classification for Britain and Ireland. Version 04.05 (internet version: www.jncc.gov.uk/MarineHabitatClassification). Joint Nature Conservation Committee, Peterborough.

CONNOR D., DALKIN M., HILL T., HOLT R. & SANDERSON W., 1997. Marine Nature Conservation Review: marine biotope classification for Britain and Ireland. Volume 2. Sublittoral biotopes. Version 97.06. Joint Nature Conservation Committee, Peterborough. pp 448.

CRISP D.J. 1964. The effects of the severe winter of 1962-63 on marine life in Britain. Journal of Animal Ecology 33: 165-210.

CRISP D.J., FISCHER-PIETTE E., 1959. Repartition des principales especes intercotidales de la cote Atlantique Francaise en 1954–1955. Annales De L Institut Oceanographique 36: 275–387.

CUNNINGHAM P.N., HAWKINS S.J., JONES H.D., BURROWS M.T., 1984. The geographical distribution of *Sabellaria alveolata* (L.) in England, Wales and Scotland, with investigations into the community structure of, and the effects of trampling on *Sabellaria alveolata* colonies. Report to the Nature

Conservancy Council from the Department of Zoology, Manchester University, Manchester. NCC report No. HF3/11/22

DANKERS N., 1993. Integrated estuarine management - obtaining a sustainable yield of bivalve resources while maintaining environmental quality. In: DAME R. R. (ed) Bivalve filter feeders in estuarine and ecosystem processes.. Springer, Berlin, 479-511.

DANKERS N., BRINKMAN A.G., MEIJBOOM A., DIJKMAN E., 2001. Recovery of intertidal mussel beds in the Waddensea: use of habitat maps in the management of the fishery. Hydrobiologia 465:21–30

DARE P.J., 1976. Settlement, growth and production of the mussel, *Mytilus edulis* L., in Morecambe Bay, England. Fishery Investigations, Ministry of Agriculture, Fisheries and Food. pp 25. Original reference not seen. Cited by Tyler-Walters, H. (2008). *Mytilus edulis*. Common mussel. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom.

DAVENPORT J. & KJORSVIK E., 1982. Observations on a Norwegian intertidal population of the horse mussel *Modiolus modiolus* (L.). Journal of Molluscan Studies, 48: 370-371.

DAVIES A.J., LAST K.S., ATTARD K., HENDRICK V.J., 2009. Maintaining turbidity and current flow in laboratory aquarium studies, a case study using *Sabellaria spinulosa*. Journal of Experimental Marine Biology and Ecology 370:35-40

DAY J.W., SCARTON F., RISMONDO A., ARET D., 1998. Rapid Deterioration of a Salt Marsh in Venice Lagoon, Italy. Journal of Coastal Research, 14: 583-590.

DENNY M.W., 1983. A simple device for recording the maximum force exerted on intertidal organisms. Limnology and Oceanography 28, 1269-1274

DEVON BAP, 2009. Grazing marsh. Devon Biodiversity and Geodiversity Action Plan. Available at: http://www.devon.gov.uk/dbap-freshwater-grazing.pdf

DE VRIES M.B., BOUMA T.J., VAN KATWIJK M.M., BORSJE B.W., VAN WESENBEECK B.K., 2007. Biobouwers van de kust. Report Z4158. WL|Delft Hydraulics, Delft, The Netherlands.

DIONNE, J.-C., 1985. Tidal marsh erosion by Geese, St. Lawrence estuary, Québec. Géographie physique et Quaternaire, 39: 99-105.

DOODY J.P., 1982. Sea defence and nature conservation: threat or opportunity. Aquat Conserv Mar Freshw Ecosyst 2: 275-283

DUBOIS S., COMMITO J.A., OLIVIER F., & RETIERE C., 2006. Effects of epibionts on *Sabellaria alveolata* (L.) biogenic reefs and their associated fauna in the Bay of Mont Saint-Michel. *Estuarine, Coastal and Shelf Science,* 68: 635-646.

DUBOIS S., RETI S., et al., 2002. Biodiversity associated with *Sabellaria alveolata* (Polychaeta: Sabellariidae) reefs: effects of human disturbances. Journal of the Marine Biological Association of the UK 82(05): 817-826.

ECKELBARGER K.J., 1978. Metamorphosis and settlement in the Sabellariidae. In: Chai, F.-S. & Rice, M. (Eds.). *Settlement and Metamorphosis of Marine Invertebrate Larvae. Proceedings of the Symposium on Settlement and Metamorphosis of Marine Invertebrate Larvae, American Zoological Society Meeting. Totonto, Ontario, Canada December 27-28, 1977*. Elsevier, New York: pp. 145-164.

ENO N.C., CLARK R.A., & SANDERSON W., Editors., 1997. *Non-native marine species in British waters: a review and directory*. Joint Nature Conservation Committee, Peterborough. pp. 152.

ERWIN D. G., PICTON B. E., CONNOR D. W., HOWSON C. M., GILLEECE P., & BOGUES M. J. 1986. The Northern Ireland sublittoral survey. Ulster Museum, Belfast. 127pp. + appendices.

EUROPEAN COMMISSION, 2007. Interpretation Manual of European Union Habitats - EUR27.

FAGER E.W., 1964. Marine Sediments: Effects of a Tube-Building Polychaete. Science 143(3604): 356-358.

FEAGIN R. A., LOZADA-BERNARD S. M., RAVENS T. M., MÖLLER I., YEAGERE K. M., BAIRD A. H., 2009. Does vegetation prevent wave erosion of salt marsh edges? PNAS 106: 10109–10113

FEY F., DANKER N., STEENBERGEN J., & GOUDSWAARD K., 2010. Development and distribution of the non-indigenous Pacific oyster (*Crassostrea gigas*) in the Dutch Wadden Sea. *Aquaculture International,* 18: 45-59.

FLEMMING B., & DELAFONTAINE M., 1994. Biodeposition in a juvenile mussel bed of the east Frisian Wadden Sea (Southern North Sea). Aquatic Ecology, 28: 289-297

FOSTER-SMITH R.L., HENDRICK V.J., 2003. *Sabellaria spinulosa* reef in The Wash and North Norfolk cSAC and its approaches: Part III, Summary of knowledge, recommended monitoring strategies and outstanding research requirements. English Nature Research Reports Number 543.

FOSTER-SMITH R.L., 2001a. Report of the field survey for the 2001 *Sabellaria spinulosa* project. A report for the Eastern Sea Fisheries Joint Committee and English Nature. pp 45.

FOSTER-SMITH R.L., 2000. Establishing a monitoring baseline for the Wash subtidal sandbanks. pp 51.

FRANK D.A., KUNS M.M., GUIDO D.R., 2002. Consumer control of grassland and plant production. Ecology 83:602

FROST M.T., LEAPER R., MIESZKOWSKA N., MOSCHELLA P., MURUA J., SMYTH C., HAWKINS S. J., 2005. Recovery of a Biodiversity Action Plan Species in Northwest England: possible role of climate change, artificial habitat and water quality amelioration. A report submitted to English Nature, Spring 2004.

GRACE J.B., FORD M.A., 1996. The potential impact of herbivores on the susceptibility of the marsh plant *Sagittaria lancifolia* to saltwater intrusion in coastal wetlands. Estuaries 19: 13–20

GARCIA, E. G., & RAGNARSSON, S. A. 2007. Impact of scallop dredging on macrobenthic communities in Breidafjordur, West Iceland. In: GARCIA, E. G., RAGNARSSON, S.A,, STEINGRIMSSON S. A, NAEVESTADD., HARALDSON H. P., FOSSA J. H., TENDAL, O. S,, & ERIKSSON H. (eds) Bottom Trawling and Scallop Dredging in the Arctic: Impacts of fishing on non-target species, vulnerable habitats and cultural heritage. Nordic Council of Ministers, Copenhagen, Chapter 2.2.

GEORGE C., & WARWICK R., 1985. Annual production in a hard-bottom reef community. *Journal of the Marine Biological Association of the United Kingdom,* 65: 713-735.

GOUDSWAARD P.C., JANSEN J.M.J., VAN ZWEEDEN C., KESTELOO J.J., VAN STRAALEN M.R., 2007. Het mosselbestand en het areaal aan mosselbanken op de droogvallende platen in de Waddenzee in het voorjarr van 2007. Wageningen IMARES, December 2007.

GRAM R., 1968. A Florida Sabellaridae reef and its effect on sediment distribution. J. Sediment. Petrol. 38: 863- 868.

GRAY A.P., LUCAS I.A.N., SEED R., and RICHARDSON C.A., 1999 *Mytilus edulis chilensis* infested with *Coccomyxa parasitica* (Chlorococcales, Coccomyxaceae). *Journal of Molluscan Studies*, 65, 289–294.

GRIME J.P., 2001. Plant Strategies, vegetation processes, and ecosystem properties. 2nd ed, J Wiley & Sons, Chichester.

GRUET Y., BODEUR Y., 1995. Ecological conditions of modern sabellarian reefs development: geological implications. Publication du Service Géologique du Luxembourg, XXIX, 73-80

GRUET Y., 1986. Spatio-temporal changes of Sabellarian reefs built by the sedentary polychaete Sabellaria alveolata (Linn6) P.S.Z.N.I. Mar. Ecol 7(4): 303-319.

GRUET Y., 1982. Recherches sur l'écologie des récifs d'Hermelles édicés par l'Annélide Polychète *Sabellaria alveolata* (Linné), Université des Sciences et Techniques, Nantes, France. PhD

GUTIERREZ J.L., JONES C.G., STRAYER D.L., IRIBARNE O.O., 2003. Mollusks as ecosystem engineers: the role of shell production in aquatic habitats. Oikos 101:79–90

HAGMEIER A., KANDLER R., 1927. Neue Untersuchungen im nordfriesischen Wattenmeer und auf den fiskalischen Austernbänken.- Wiss. Meeresunters. (Abt. Helgoland) 16, 1-90.

HENDRICK V.J. & FOSTER-SMITH R.L., 2006. *Sabellaria spinulosa* reef: a scoring system for Revaluating 'reefiness' in the context of the Habitats Directive. Journal of the Marine Biological Association of the UK, 86, 665-677.

HENDRICK V. J., FOSTER-SMITH R. L. & DAVIES A. J., 2011. Biogenic Reefs and the Marine Aggregate Industry. Marine ALSF Science Monograph Series No. 3. MEPF 10/P149. (Edited by R. C. NEWELL & J. MEASURES). 60pp. ISBN: 978 0 907545 46 0.

HESLENFELD P., JUNGERIUS P.D., KLIJN J.A, 2004. European coastal dunes: ecological values, threats, opportunities and policy development. Ecological Studies, Vol 171. Coastal dunes, ecology and conservation. MARTINEZ M. L , PSUTY N.P., Eds.

HEURTEFEUX H., GROSSET S., RICHARD P., SIRE E., 2007. Restoring a highly damaged site: Canet-en-Roussillon (Western French Mediterranean coast). ICCD, 2007. Montpellier-France. 7pp.

HEURTEFEUX H., GROSSET S., VALANTIN P.-Y., 2007. Une approche alternative de la gestion des risques côtiers, l'exemple de la Petite Camargue. Territoires en mouvement 2007-1. Les risques côtiers. 11pp.

HEURTEFEUX H., and RICHARD P., 2005. Vingt ans de protection des dunes littorals par des techniques douces : bilan et perspectives à travers quelques exemples en Méditerranée française. Milieux littoraux, nouvelles perspectives d'étude. Paris-France. p. 143-166

HEURTEFEUX H., SAUBOUA P., LANZELLOTTI P., BICHOT A., 2011, in press. Coastal risk management modes : The managed realignment as a risks conception more integrated. Montpellier-France

HIGLER B., DANKERS N., SMAAL A.,DE JONGE V.N., 1998. Evaluatie van de ecologische effecten van het reguleren van schlpdievisserij in Waddenzee en Delta op bodemorganismen en vogels. In: VAN DIJK J.J. and R. HEILING (eds.) Structuurnota Zee- en Kustvisserij, van de maatregelen in de kustvisserij gedurende de erste fase (1993–1997). Appendix 5, pp. 17.

HOLT T.J., REES E.I., HAWKINS, S.J., SEED, R., 1998. Biogenic Reefs (volume IX). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 170 pp.

HOLT T., HARTNOLL R. & HAWKINS S., 1997. Sensitivity and vulnerability to man-induced change of selected communities: intertidal brown algal shrubs, *Zostera* beds and *Sabellaria spinulosa* reefs. English Nature Research Reports. No. 234. pp 97.

HOLT T.J., & SHALLA S.H.A., 1997. Pre- and post-drilling survey of block IOM 112/19, A report to Elf Enterprise Caledonia Ltd. by Port Erin Marine Laboratory, University of Liverpool. Unpublished work.

HOWARD R.J., MENDELSSOHN I.A., 2000. Structure and composition of oligohaline marsh plant communities exposed to salinity pulses. Aquat Bot 68: 143–164

HOWSON C., CONNOR D., HOLT R., 1994. The Scottish sealochs – an account of surveys undertaken for the Marine Nature Conservation Review. Joint Nature Conservation Committee Report, No. 164.

HUGHES R.G., PARAMOR O.A.L., 2004. On the loss of saltmarshes in south-east England: methods for their restoration. Journal of Applied Ecology, 41: 440-448.

JACKSON A. & HISCOCK K., 2008. *Sabellaria spinulosa*. Ross worm. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from: http://www.marlin.ac.uk/species/Sabspi.htm

JACKSON J., 1977. Competition on marine hard substrata: the adaptive significance of solitary and colonial strategies The American Naturalist*,* 111: 743-767.

JASIM A.K., & BRAND A.R., 1989. Observations on the reproduction of *Modiolus modiolus* in Isle of Man waters. Journal of the Marine Biological Association of the UK*,* 69: 373-385.

JENSEN R.A., 1992. Marine bioadhesive: role for chemosensory recognition in a marine invertebrate. *Biofouling,* 5: 177-193.

JESSOP R., & STOUTT J., 2006. Broad scale *Sabellaria spinulosa* distribution in the central Wash (Southern North Sea), as predicted with the acoustic ground discriminating system (A.G.D.S) RoxannTM. Draft report by the Eastern Sea Fisheries Joint Committee for English Nature. pp 26.

JONES L.A., HISCOCK K., CONNOR D.W., 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

JONES L., 1999. Habitat Action Plan: *Sabellaria spinulosa* reefs. English Nature. pp 4.

JONES C.G., LAWTON J.H., & SHACHAK M., 1999. Positive and negative effects of organisms as physical ecosystem engineers. Ecology 78(7): 1946-1957.

KENDALL M.A., & BEDFORD M.L., 1987. Reproduction and recruitment in the barnacle *Chthamalus montagui*at Aberystwyth (mid-Wales). Marine Ecology Progress Series 38: 305-308.

KIEHL K., EISCHEID I., GETTNER S., WALTER J., 1996. Impact of different sheep grazing intensities on salt-marsh vegetation in northern Germany. Journal of Vegetation Science, 7, 99–106

KIEHL K., SCHRÖDER H., STOCK M., 2007. Long-term vegetation dynamics after land-use change in Wadden Sea salt marshes. Coastline Reports, 7: 17-24.

KIRTLEY D.J., 1992. Built to last. Worm reefs. A feat of natural engineering. *Florida Oceanographic Magazine,* l3: 12-19.

KIRTLEY D.J., 1966. Intertidal reefs of Sabellariidae (Annelida polychaeta) along the coasts of Florida. Masters thesis. The Florida State University. Tallahassee, Florida. 104 pp. Original reference not seen. Cited by Drake, C.A., McCarthy, D.A. & von Dohlen, C.D. (2007). Molecular relationships and species divergence among *Phragmatopoma* spp. (Polychaeta: Sabellaridae) in the Americas. *Marine Biology* 150(3): 345-358.

KIRWAN M., TEMMERMAN S., 2009. Coastal marsh response to historical and future sea-level acceleration. Quaternary Science Reviews, 28: 1801-1808.

KLEIN R.J.T., BATEMAN I.J., 1998. The recreational value of Cley marshes nature reserve: An argument against managed Retreat? Water and Environment Journal 12: 280-285

KOCH E.W., BARBIER E.B., SILLIMAN B.R., REED D.J., PERILLO G.M.E., HACKER S.D., GRANEK E.F., PRIMAVERA J.H., MUTHIGA, N., POLASKY S., HALPERN B.S., KENNEDY C.J., KAPPEL C.V., WOLANSKI E. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment* 7(1):29-37.

KRISTENSEN P.S., 1995. Aerial surveys, biomass estimates, and elimination of the mussel population (*Mytilus edulis* L.), in the Danish Wadden Sea, 1991±1994. ICES C.M. 1995/K:44, 22 pp.

LARSONNEUR C. 1994. The Bay of Mont-Saint-Michel: A sedimentation model in a temperate macrotidal environment, *Senckenbergiana maritima*, 24, 3-63.

LAST K.S., HENDRICK V.J., BEVERIDGE C.M. & DAVIES A.J., 2011. Measuring the effects of suspended particulate matter and smothering on the behaviour, growth and survival of key species found in areas associated with aggregate dredging. Report for the Marine Aggregate Levy Sustainability Fund, Project MEPF 08/P76. 69 pp.

LIMPENNY D.S., FOSTER-SMITH R.L., EDWARDS T.M., HENDRICK V.J., DIESING M., EGGLETON J.D., MEADOWS W.J., CRUTCHFIELD Z., PFEIFER S., & REACH I.S., 2010. Best methods for identifying and evaluating *Sabellaria spinulosa* and cobble reef. Aggregate Levy Sustainability Fund Project MAL0008. Joint Nature Conservation Committee, Peterborough. pp 134.

LINDENBAUM C., BENNELL J., REES E., MCCLEAN D., COOK W., WHEELER A., SANDERSON W., 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea: I. Seabed mapping and reef morphology. Journal of the Marine Biological Association of the UK, 88, 133- 141.

LINKE O., 1951. Neue Beobachtungen uber Sandkorallen-Riffe in der Nordsee, *Natur u. Volk.*, 81, 77-84.

LIVINGSTONE D.R., & PIPE R.K., 1992. Mussels and environmental contaminants: molecular and cellular aspects. In: Gosling, E. (Ed.). *The mussel Mytilus: ecology, physiology, genetics and culture*. Developments in Aquaculture and Fisheries Science 25. Elsevier Press, Amsterdam: pp. 425- 510.

LUTZ R.A., & KENNISH M.J., 1992. Ecology and morphology of larval and early postlarval mussels. In: Gosling, E. (Ed.). *The mussel Mytilus: ecology, physiology, genetics and culture*. Developments in Aquaculture and Fisheries Science 25. Elsevier Press, Amsterdam: pp. 53-86.

MAGORRIAN B.H., & Service, M., 1998. Analysis of underwater visual data to identify the impact of physical disturbance on horse mussel *(Modiolus modiolus*) beds. *Marine Pollution Bulletin,* **36**: 354- 359.

MAIR J. M., MOORE C. G., KINGSTON P. F. & HARRIES D. B., 2000. A review of the status, ecology and conservation of horse mussel Modiolus modiolus beds in Scotland. Scottish Natural Heritage, Edinburgh (Commissioned Report F99PA08).

MARCHAND Y., CAZOULAT R., 2003. Biological reef survey using spot satellite data classification by cellular automata method - Bay of Mont Saint-Michel (France). Computers & Geosciences 29, 413-421

MARTINEZ M.L. and PSUTY N.P., 2004. Coastal Dunes. Ecology and Conservation. Ecological Studies, 171: 386pp

MCCARTHY D., 2001. Life-history patterns and the role of disturbance in intertidal and subtidal populations of the polychaete *Phragmatopoma lapidosa lapidosa* (Kinberg, 1867) in the tropical Western Atlantic. PhD Thesis. Kings College, University of London. Original reference not seen. Cited by Drake, C.A., McCarthy, D.A. & von Dohlen, C.D. (2007). Molecular relationships and species divergence among *Phragmatopoma* spp. (Polychaeta: Sabellaridae) in the Americas. Marine Biology **150**(3): 345-358.

MCCARTHY D., YOUNG C. & EMSON R., 2003. Influence of wave induced disturbance on seasonal spawning patterns in the sabellariid polychaete *Phragmatopoma lapidosa*. Marine Ecological Progress Series, 256: 123-133.

MCGRATH D., KING P., & GOSLING E., 1988. Evidence for the direct settlement of *Mytilus edulis* larvae on adult mussel beds. Marine Ecological Progress Series, 47: 103-106.

MCKAY D., 1994. Unravelling the choreography of contaminant kinetics: approaches to quantifying the uptake of chemicals by organisms. In: J.L. Hamelink, P.F. Landrum, H.L. Bergman and W.H. Benson (Editors), Bioavailability: Physical, Chemical, and Biological Interactions, Lewis Publisher Inc., Chelsea, MI., pp. 17l-l77.

MCKEE K.L., CHERRY J.A., 2009. Hurricane Katrina sediment slowed elevation loss in subsiding brackish marshes of the Mississippi river delta. Wetlands 29: 2–15

MEYLES E.W., WILLIAMS A.G., TERNAN J.L., ANDERSON J.M., DOWD J.F., 2006. The influence of grazing on vegetation, soil properties and stream discharge in a small Dartmoor catchment, southwest England, UK. Earth Surface Processes and Landforms 31, 622-631

MICHAELIS H., 1978. Recent biological phenomena in the German Waddensea. *Symposium on North Sea fish stocks – recent changes and their causes.*, Aarhus (Denmark).

MIESZKOWSKA N., KENDALL M.A., HAWKINS S.J., Leaper R., Williamson P., Hardman-Mountford N.J., SOUTHWARD A.J., 2006. Changes in the range of some common rocky shore species in Britain - a response to climate change? Hydrobiologia, 555, 241-51

MILOTIC T., ERFANZADEH R., PETILLON J., MAELFAIT J.P., Hoffmann M., 2010. Short-term impact of grazing by sheep on vegetation dynamics in a newly created salt-marsh site. Grass and Forage Science 65, 121-132

MÖLLER I., SPENCER T., FRENCH J.R., LEGGET D.J., DIXON M., 1999. Wave transformation Over Salt Marshes: A Field and Numerical Modelling Study from North Norfolk, England. Estuarine, Coastal and Shelf Science 49, 411–426

MÖLLER I., SPENCER T., 2002. Wave dissipation over macro-tidal saltmarshes: Effects of marsh edge typology and vegetation change. Journal of Coastal Research SI36, 506-521

MÖLLER I., 2006. Quantifying saltmarsh vegetation and its effect on wave height dissipation: Results from a UK East coast saltmarsh. Estuarine Coastal and Shelf Science, 69: 337-351.

MOULIS D., and BARBEL P., 1999. Restauration des dunes. Réhabilitation et gestion des dunes littorals Méditerranéennes Françaises. Collection : Manuels et Méthodes. BRGM Ed., 75-91.

MULTER H.G., MILLIMAN J.G.,1967. Geologic aspects of sabellarian reefs, Southeastern Florida. Bulletin of Marine Science 17: 257- 267.

NAYLOR L.A., VILES H.A., 2000. A temperate reef builder: an evaluation of the growth, morphology and composition of *Sabellaria alveolata* (L.) colonies on carbonate platforms in South Wales. In: E.R. Insalaco, P.W. Skelton and T.J. Palmer, Editors, *Carbonate platform systems: components and interactions*, *Spec. Publ.-Geol. Soc. Lond.* vol. 178 (2000), pp. 9–19.

NAYLOR L.A., 2001. An assessment of the links between biogenic processes and shore platform geomorphology. Glamorgan Heritage Coast, South Wales, UK. Thesis, D. Phil., University of Oxford.

NEHLS G., DIEDERICH S., THIELTGES D., & STRASSER M., 2006. Wadden Sea mussel beds invaded by oysters and slipper limpets: competition or climate control? *Helgoland Marine Research,* 60: 135-143.

NEUHAUS R., STELTER T., KIEHL K., 1999. Sedimentation in Salt Marshes Affected by Grazing Regime, Topographical Patterns and Regional Differences. Senckenbergiana maritima 29,113-116

NICHOLLS R.J., WILSON T., 2001. Chapter five. Integrated impacts on coastal areas and river flooding. In: Holman IP, Loveland PJ (Eds), Regional Climate Change Impact and Response Studies in East Anglia and

North West England (RegIS). Final Report of MAFF project no. CC0337. (downloadable at www.ukcip.org.uk).

O'CONNELL J., 2008. Coastal dune Protection & Restoration. Using « Cape » American Beachgrass & Fencing. Marine Extension Bulletin December 2008. 20pp.

O'CONNOR N.E., CROWE T.P., 2007. Biodiversity among mussels: separating the influence of sizes of mussels from the ages of patches. Journal of the Marine Biological Association of the United Kingdom 89: 551-557.

OOST A.P., 1995. The influence of biodeposits of the blue mussel *Mytilus spp.* on fine- grained sedimentation in the temperate-climate Dutch Wadden Sea. Geologica Ultraiectina, 126, 359-400.

OSPAR, 2010. Quality Status Report 2010. OSPAR Commission. London. 176 pp.

PENNINGS S.C., CALLOWAY R.M., 1992. Salt marsh plant zonation: the relative importance of competition and physical factors. Ecology 73: 681-690

PERKINS E.J. 1981. Studies in the distribution and biological impact of the effluent released by Thames Board Mills Ltd., Siddick. 5. The bottom fauna of Maryport Roads and Allonby Bay 1962-1978. Scientific Report of the Cumbria Sea Fisheries Committee, Rep. No. 81.

PETERSON J.E., BALDWIN A.H., 2004. Seedling emergence from seed banks of tidal freshwater wetlands: response to inundation and sedimentation. Aquat Bot 78: 243–254

PIAZZA B.P., BANKS P.D., LA PEYRE M.K. 2005. The Potential for Created Oyster Shell Reefs as a Sustainable Shoreline Protection Strategy in Louisiana, Restoration Ecology 13:499– 506

POPPE G., GOTO Y., 1993. *European seashells*. Volume:2 (Scaphopoda, Bivalvia, Cephalopoda). Conchbooks, Haekenheim. 221 pp.

REISE K., 2002. Sediment mediated species interactions in coastal waters. Journal of Sea Research 48(2): 127-141.

REISE K., 2005. Coast of change: habitat loss and transformations in the Wadden Sea. Helgoland Marine Research 59(1): 9-21.

REISE K., SCHUBERT A., 1987. Macrobenthic turnover in the subtidal Wadden Sea: The Norderaue revisited after 60 years. Helgolander Meeresuntersuchungen, 41, 69-82.

RIESEN W., REISE K., 1982. Macrobenthos of the subtidal Wadden Sea: Revisited after 55 years, Helgolander Meeresuntersuchungen, 35, 409-423.

ROBERTS D., & MCKENZIE J.D., 1983. Utilisation of mollusc resources in N. Ireland. Journal of Molluscan Studies, 49: 162-166.

ROLLET C., BONNOT-COURTOIS C., FOURNIER J., 2005. Cartographie des habitats benthiques médiolittoraux à partir des orthophotographies littorales. Fiche technique - Projet REBENT FT13-2005-01, Ifremer, Brest. 18pp

SABATIER F., 2001. Fonctionnement et dynamiques morpho-sédimentaires du littoral du delta du Rhône. Thèse. Université Aix-Marseille III.272pp.

SAMUELS J.S., MASON C.F., 1997. Ecology of eutrophic waterbodies in a coastal grazing marsh. Hydrobiologia 346: 203–214

SCHAFER W., 1972. *Ecology and Palaeoecology of Marine Environments*. Translation of *Aktuo-paläontologie nach Studien in der Nordsee*. University of Chicago Press, Chicago. pp 568.

SCHEFFER M., CARPENTER S., FOLEY J.A., FOLKER C., WALKER B., 2001. Catastrophic shifts in ecosystems. Nature, 413: 591-596.

SCHEFFER M., BASCOMPTE J., BROCK W.A., BROVKIN V., CARPENTER S., DAKOS V., HELD H., VAN NES E.H., RIETKERK M., SUGIHARA G., 2009. Early-warning signals for critical transitions. Nature, 461: 53-59.

SCHWEINITZ E., & LUTZ R., 1976. Larval development of the northern horse mussel, *Modiolus modiolus* (L.), including a comparison with the larvae of *Mytilus edulis* L. as an aid in planktonic identification. Biological Bulletin*,* 150: 348-360.

SEED R., & BROWN R.A., 1977. Comparison of reproductive cycles of *Modiolus modiolus* (L), *Cerastoderma* (= *Cardium*) *edule* (L), and *Mytilus edulis* L in Strangford Lough, Northern Ireland. Oecologia*,* 30: 173-188.

SEED R., 1969. The ecology of *Mytilus edulis* L. (Lamellibranchiata) on exposed rocky shores. Oecologia*,* 3: 317-350

SEED R., 1976. Ecology. In: Bayne, B. (Ed.). *Marine mussels: their ecology and physiology*. International Biological Programme 10. Cambridge University Press, Cambridge: pp. 13-66.

SEED R., & BROWN R., 1978. Growth as a strategy for survival in two marine bivalves, *Cerastoderma edule* and *Modiolus modiolus*. Journal of Animal Ecology*,* 47: 283-292.

SEED R. & SUCHANEK T.H., 1992. Population and community ecology of *Mytilus*. In: Gosling, E. (Ed.). *The mussel Mytilus: ecology, physiology, genetics and culture*. Developments in Aquaculture and Fisheries Science 25. Elsevier, Amsterdam: pp. 87-170.

SERVICE M., MAGORRIAN B. H., 1997. The extent and temporal variation of disturbance of epibenthic communities in Strangford Lough, Northern Ireland. Journal of the Marine Biological Association of the United Kingdom, 77, 1151-1164.

SHARPE P.J., BALWIN A.H., 2009. Patterns of Wetland Plant Species Richness Across Estuarine Gradients of Chesapeake Bay. Wetlands 29: 225-235

SOOT-RYEN T., 1955. A report on the family Mytilidae. Allan Hancock Pacific Expedition, 20, 1-154

STEWART R.J., WEAVER J.C., MORSE D.E. & WAITE J.H., 2004. The tube cement of *Phragmatopoma californica*: a solid foam. Journal of Experimental Biology, 207: 4727-4734.

SUKHOTIN A.A., STRELKOV P.P., MAXIMOVICH N.V. & HUMMEL H., 2007. Growth and longevity of *Mytilus edulis* (L.) from northeast Europe. *Marine Biology Research,* **3**: 155 – 167.

TARAMELLI-RIVOSECCHI E., 1961. Observazioni sulle biocensoi del bianco a *Sabellaria* di Lavinio. Rendiconti – Academia nazionale delle scienze detta dei XL. Memoire di Matematica $: 147-154

TAYLOR P.M., PARKER J.G., 1993. An Environmental Appraisal: The Coast of North Wales and North West England, Hamilton Oil Company Ltd, 80 pp.

TEBBLE N., 1966. British bivalve seashells. Natural History Museum, London. pp 212.

THIELTGES D.W., STRASSER M., REISE K., 2003. The American slipper limpet *Crepidula fornicata* (L.) in the northern Wadden Sea 70 years after its introduction. Helgoland Marine Research, 57: 27–33.

THIELTGES D.W., 2005a. Impact of an invader: epizootic American slipper limpet *Crepidula fornicata* reduces survival and growth in European mussels. Marine Ecology Progress Series 286:13–19

THIELTGES D.W., 2005b. Benefit from an invader: American slipper limpet *Crepidula fornicata* reduces star fish predation on basibiont European mussels. Hydrobiologia 541(1): 241-244.

TORRRES R., FAGHERAZZI S., VAN PROOSDIJ D., HOPKINSON D., 2006. Salt marsh geomorphology: Physical and ecological effects on landform. Estuarine, Coastal and Shelf Science 69, 309-310

TSOAR H., 2005. Sand dunes mobility and stability in relation to climate. Physica A, 357, 50-56.

TYLER-WALTERS H., 2008. *Mytilus edulis*. Common mussel. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from www.marlin.ac.uk/speciesfullreview. php?speciesID=3848.

TYLER-WALTERS H., 2007. *Modiolus modiolus*. Horse mussel. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 01/05/2011]. Available from: http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3817

TZATZANIS M., WRBKA T., SAUBERER N., 2003. Landscape and vegetation responses to human impact in sandy coasts of Western Crete, Greece. J. Nat. Conserv. 11, 187-195

VAN DE KOPPEL J., VAN DER WAL D., BAKKER J.P., HERMAN P.M.J., 2005. Self-Organization and Vegetation Collapse in Salt Marsh Ecosystems. The American Naturalist, 165: E1-12.

VAN DER MEULEN F. and SALMAN A.H.P.M., 1993. Gestion des dunes côtières de Méditerranée. The first International Conference on the Mediterranean Coastal Environment. 167-183.

VAN DER WAL D., PYE K., 2004. Patterns, rates and possible causes of saltmarsh erosion in the Greater Thames area (UK). Geomorphology, 61: 373-391.

VAN DER WAL D., WIELEMAKER-VAN DEN DOOL A., HERMAN P.M.J., 2008. Spatial patterns, rates and mechanisms of saltmarsh cycles (Westerschelde, The Netherlands). Estuarine, Coastal and Shelf Science, 76: 357-368.

VAN HULZEN J., VAN SOELEN J., HERMAN P.M.J., BOUMA T.J., 2006. The significance of spatial and temporal patterns of algal mat deposition in structuring salt marsh vegetation. J Veget Sci 17: 291-298

VAN KATWIJK M.M., 2003. Reintroduction of eelgrass (*Zostera marina* L.) in the Dutch Wadden Sea, a research overview and management vision. p.173-197. In: Challenges to the Wadden Sea area. Wolff W.J., K. Essink, A. Kellermann and M.A. van Leeuwe (Eds). Proceedings of the 10th International Scientific Wadden Sea Symposium.

VAN LEEUWEN B., AUGUSTIJN D.C.M., VAN WESENBEEK B.K., HULSCHER S.J.M.H., & DE VRIES M.B., (2010). Modeling the influence of a young mussel bed on fine sediment dynamics on an intertidal flat in the Wadden Sea. Ecological Engineering 36(2): 145-153.

VAN WESENBECK B.K., VAN DE KOPPEL J., HERMAN P.M.J., BERTNESS M.D., VAN DER WAL D., BAKKER J. P., BOUMA T.J., 2008. Potential for Sudden Shifts in Transient Systems: Distinguishing Between Local and Landscape-Scale Processes. Ecosystems 11, 1133-1141

VEALE L.O., HILL A.S., HAWKINS S.J., BRAND A.R., 2000. Effects of long-term physical disturbances by commercial scallop fishing on subtidal epifaunal assemblages and habitats. Marine Biology, 137: 325-337.

VERA F.W.M., 2000. Grazing Ecology and Forest History. CABI Publishing, Wallingford, UK.

VINCHON C., BALOUIN Y., IDIER D., GARCIN M., MALLET C., 2008. La réponse du trait de côte au changement climatique : Evolution des risques côtiers en Aquitaine et en Languedoc-Roussillon dans le siècle à venir. The littoral : challenge, dialogue, action. Lille-France. 11pp.

VORBERG R., 2000. Effects of the shrimp fisheries on reefs of *Sabellaria spinulosa* (Polychaeta). ICES Journal of Marine Science, 57, 1416–1420.

WARREN P., 1973. The fishery for the pink shrimp *Pandalus montagui* of the Wash. Laboratory Leaflet (New Series) No. 28. Ministry of Agriculture, Fisheries and Food, Lowestoft. pp. 46.

WARREN P.J., SHELDON R.W., 1967. Feeding and migration patterns of the Pink Shrimp *Pandalus montagui*, in the estuary of the River Crouch, England, Journal of the Fisheries Research Board of Canada, 24, 569-580.

WEHRMANN A, MARKERT A, SCHMIDT A., 2007 Miesmuschelbank: ein verlorener Lebensraum? Die Einwanderung der Pazifischen Auster in das Wattenmeer und ihre Folgen. Natur- und Umweltschutz 6(1):10–14

WIDDOWS J., & DONKIN P., 1992. Mussels and environmental contaminants: bioaccumulation and physiological aspects. In: Gosling, E. (Ed.). *The mussel Mytilus: ecology, physiology, genetics and culture*. Developments in Aquaculture and Fisheries Science 25. Elsevier Press, Amsterdam: pp. 383-424.

WIDDOWS J., & SHICK J.M., 1985. Physiological responses of *Mytilus edulis* and *Cardium edule* to aerial exposure. Marine Biology*,* 85: 217-232.

WIDDOWS J., BRINSLEY M.D., 2002. Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. Journal of Sea Research 48: 143–156.

WIDDOWS J., DONKIN P., STAFF F.J., MATTHIESSEN P., LAW R.J., ALLEN Y.T., THAIN J.E., ALLCHIN C.R., JONES B.R. 2002. Measurement of stress effects (scope for growth) and contaminant levels in mussels (*Mytilus spp.*) collected from the Irish Sea. Marine Environmental Research, 53, 327–356

WILDISH D.J., FADER G. & PARROTT D., 2009. A model of horse mussel reef formation in the Bay of Fundy based on population growth and geological processes. Atlantic Geology*,* 45, 157- 170.

WILSON D.P., 1929. The larvae of the British Sabellarians. Journal of the Marine Biological Association of the United Kingdom, 15: 221-269.

WILSON D.P., 1968. The settlement behaviour of the larvae of *Sabellaria alveolata*. Journal of the Marine Biological Association of the United Kingdom, 48: 387-435.

WILSON D.P., 1970. The larvae of *Sabellaria Spinulosa* and their settlement behaviour. Journal of the Marine Biological Association of the United Kingdom, 50, 33-52

WILSON D.P., 1971. *Sabellaria* colonies At Duckpool, North Cornwall, 1961-1970. Journal of the Marine Biological Association of the UK, 51, 509-580.

WILSON D.P., 1974. *Sabellaria* Colonies at Duckpool, North Cornwall, 1971–1972, With a Note for May 1973. Journal of the Marine Biological Association of the United Kingdom, 54, 393-436

WILSON D.P., 1976. *Sabellaria Alveolata* (L.) At Duckpool, North Cornwall, 1975. Journal of the Marine Biological Association of the United Kingdom, 56, 305-310

WOLTERS M., BAKKER J.P., BERTNESS M.D., JEFFERIES R.L., MÖLLER I., 2005. Saltmarsh erosion and restoration in south-east England: squeezing the evidence requires realignment. Journal of Applied Ecology, 42: 844-851.

WOOD R., 1999. *Reef Evolution*. Oxford University Press, Oxford. pp 414.

YANG S.L., LI H., YSEBAERT T., BOUMA T.J., ZHANG W.X., WANG Y.Y., LI P., LI M., DING P.X., 2008. Spatial and temporal variations in sediment grain size in tidal wetlands, Yangthe Delta: on the role of physical and biotic controls. Estuarine, Coastal and Shelf Science, 77: 657-671.

VIII. APPENDICES

VIII.1 DESCRIPTION OF COASTAL HABITATS WITHIN THESEUS STUDY SITES

VIII.1.1 INTRODUCTION TO PILOT SITES

The THESEUS project has 8 European Pilot sites: Santander spit (Gulf of Biscay,Atlantic Ocean), Gironde Estuary(Atlantic Ocean), Plymouth Sound to Exe Estuary (English channel), Scheldt estuary (North Sea), Elbe Estuary(North Sea), Po delta and adjoning coast (Mediterrenan Sea), Vistula delta plain(Baltic Sea), Varna spit (Black Sea).

This report focus on four of them: Santander, Gironde Estuary, Plymouth and Scheldt.

Plymouth pilot site

The Plymouth site is located in Devon on the English south coast. The site extends 10 km south from Exeter to the open sea at Dawlish Warren. It comprises the waters, foreshore, low-lying land, three saltmarshes and an unusual double spit across the mouth of the estuary, and the sand dunes of Dawlish Warren. The mud- and sand-flats support Eelgrass Zostera spp. and Enteromorpha beds, and contain an abundance of invertebrates including extensive Mussel *Mytilus edulis beds.* The Plymouth THESEUS site has eight main estuaries, all of which are characterised as Rias. These are noted for their steep banks and narrow widths and therefore have reduced saltmarsh areas when compared to delta type estuaries. Focus will be done on three of the 8 estuaries: Plymouth Sound, which covers the Plym estuary and a wider area of estuarine systems and coast, the Teign and the Exe estuaries.

Santander pilot site

The Bay of Santander is the largest estuary on the North coast of Spain, at the Gulf of Biscay, with an extension of 9 km long and 5 km wide. The Northern coast of Spain is divided into a series of pocket beaches and small inlets isolated between rocky headlands. Waves on the Gulf of Biscay approach mostly from the NW with a mean significant wave height of 1 m and a typical winter storm wave of 5 m. The entrance to the Bay is lined by the Sardinero beaches. The access to its interior is through a narrow channel of water between the Magdalena Peninsula, and the sandbanks of the El Puntal, a 2.5 km long series of beaches and dunes that protect the inner waters of the Bay. The natural coastal habitats of Santander focussed in this report are: salt marshes, dunes and seagrasses.

Gironde Estuary

Gironde estuary is an estuary in South-West of France located on two regions: Aquitaine and Poitou Charentes. This is a common estuary of Garonne and Dordogne Rivers. It is the largest estuary in Western Europe. It is 75 km long, up to 12 km wide. This macrotidal estuary has tide amplitude varying from 2 m to 5 m. This amplitude does not damp. On the contrary it increases from river mouth to upstream areas because of decreasing width and depth. Despite its quite strong artifialisation, mosaic of natural meadows is found in the Estuary.

Scheldt estuary

The Western Scheldt estuary is located in the SW of the Netherlands, connecting the Scheldt River in Belgium with the North Sea coast of the Netherlands. Along the Western Scheldt, the land is protected by strong dikes, as much of this land is laying well below sea level. The Western Scheldt is of major economic importance, as it provides the navigation channel for the harbour of Antwerp, which is essential fo rr the Belgium economy.

The Western Scheldt itself is a multi channel system with several tidal flats both in the middle and along the edges of the estuary. These tidal flats are inhabited by a rich diversity of benthic organisms, on which migratory birds feed. At higher elevations are tidal marshes with unique vegetation. The biology in these areas is governed by a combination of an elevational (flooding) gradient and a salinity gradient. Whereas the mouth of the Western Scheldt is saline, more inland the estuary becomes more brackish due to the influence of the Scheldt River. This salinity gradient has a major influence of the benthos composition of the tidal flats and the vegetation composition at the tidal marshes. Nearby Antwerp, is the Verdronken land of Saeftinge, which is the largest remaining brackish tidal marsh of the whole Europe. Because of its unique ecological value, the nature in the Western Scheldt is legally highly protected, thereby putting constraints on the economic development of the harbour of Antwerp.

University of Bangor partners from Northwales (UK) have also given local information from their coasts; this information will be presented here at the end of the Plymouh pilot site chapter.

Emilia Romagna Region

Bellocchio and Cesanatico are two locations along the Emilia Romagna coast, which borders the North Western Adriatic sea, from the Sacca di Goro south to Cattolica. The Emilia Romagna coast, consists of a flat alluvial sandy system, with no natural rocky reefs and almost uninterrupted except for the mouths of rivers, channels and lagoon systems, including the Po river delta. Most of these lagoons have been filled by sediments or reclaimed over historical times, and what remains today has been recognised as an internationally important wetland according to the Ramsar convention and as a Special Protected Area and/or proposed sites of Community interest, according to the EU Bird and Habitat Directives. The area is subject to moderate wave action and to a tidal range over 80 cm. Average surface sea temperatures vary between 8 °C in the winter and 24 °C in the summer, with peaks above and below these values close to the coastline. Human pressure on the coast is particularly important in Cesenatico, where intensive coastal development has taken the place of most natural habitats. Conversely Bellocchio is a State Nature Reserve and a Nature 2000 Site, and comprises a coastal lagoon separated from the sea by a thin sandbank and characterized by important halophile vegetation and avifauna.

VIII.1.2 PLYMOUTH

Salt marshes

Figure 8.1. Salt marshes and Intertidal Mud flats in the Plymouth site

In the UK, three major types of saltmarsh are recognised (Dijkema, 1987):

1) Barrier-connected saltmarshes (these marshes develop in the lee of spits or barrier islands),

2) Foreland saltmarshes (This type of marsh develop in front of sheltered, alluvial coastal pains)

3) Estuarine saltmarshes (Estuarine marshes are found where rivers gradually merge into the open sea).

There are approximately 40 species of higher plant found within British saltmarshes but more typically only 10 -20 species are found in any one marsh.

Typical species zonation

The UK typically has five main saltmarsh zones, including an upper transitional zone. These zones are defined by plant distribution and tend to be linked to tidal regime.

The five zones are:

1. Pioneer: This zone is covered by all tides except the lowest neaps and is defined as an open community with one or more of the following: *Spartina* spp., *Salicornia* spp., *Aster tripolium*.
2. Low marsh: These are covered by most tides and are generally closed communities with at least *Puccinellia maritima* and *Atriplex portulacoides* in addition to the previous species above.
3. Middle marsh: Here the zone is only covered by spring tides, is generally a closed community consisting of *Limonium* spp. and/or *Plantago* spp. as well as the above species.
4. High marsh: This zone is only covered by the highest spring tides and is generally a closed community consisting of *Festuca rubra*, *Armeria maritima* and *Elytrigia* spp. and the previous species.
5. Transition zone: This area is only inundated during occasional extreme storm events and strong tidal surges and is defined by vegetation between high marsh and the adjoining no-halophytic communities.

In areas where physical characteristics are extreme (i.e. where marshes are exposed to high wave energy), middle to high marsh can occur above the normal spring tide level (or they are restricted by sea walls) and high zone marsh may be noted as absent with transition zones seen immediately behind sea defences.

Issues that affect UK saltmarsh are currently characterised as **1)** reclamation (the alteration of marsh for cattle grazing and agriculture), **2)** isostatic adjustment (following the removal of ice sheets from the Scottish highlands) and **3)** habitat squeeze (where marshes are unable to re-form at higher landward elevations).

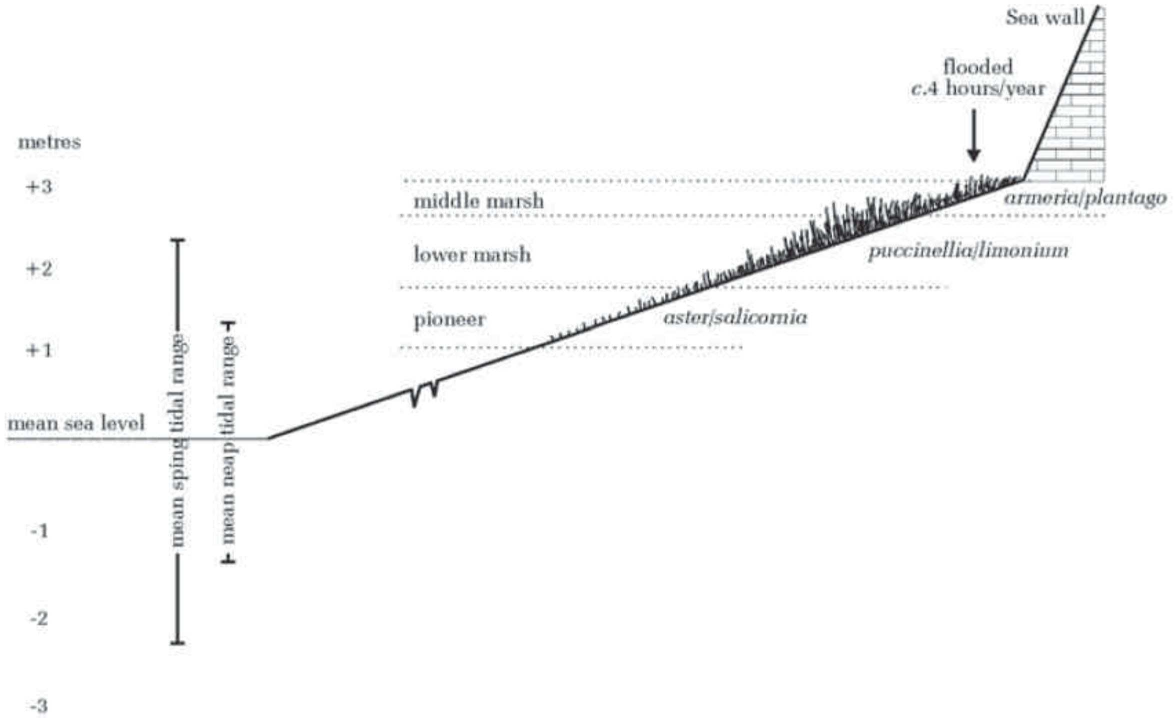
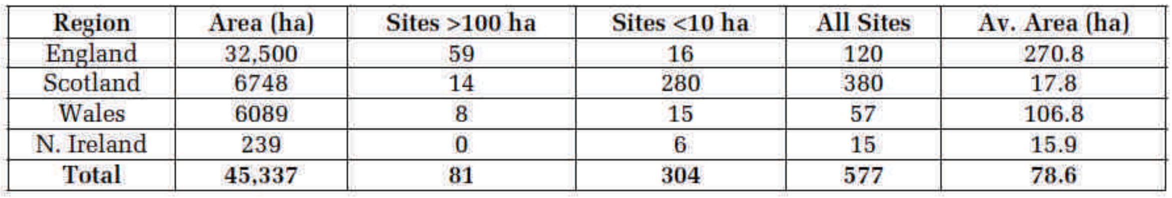


Figure 8.2. Diagrammatic representation of the tidal changes in the water level at a typical saltmarsh site, indicating the major vegetation zones

Description of management status

There are several Red Data Book vascular plant species of saltmarsh: *Atriplex pedunculata, Chenopodium chenopodioides, Euphrasia heslop-harrisonii, Hierochloe odorata, Limonium bellidifolium, L. binervosumI* spp. *anglicum* and spp. *binervosum, L. britannicum* spp. *britannicum, celticum* and *trancanalis, Limosella australis, Polygonum maritimum*.

50 species of mosses (only one of which is restricted to saltmarsh) and one liverwort have been recorded along with many michorrhizal fungi, micro-fungi and parasitic fungi. Of the 293 invertebrates found in saltmarshes 148 of these are restricted to this habitat. Vertebrates are present in two major groups, birds and fish. Saltmarshes are important for a range of species within these groups including, fish nurseries and feeding grounds and bird roosts and feeding grounds specifically for passerine and over-wintering migrant birds.



Region	Area (ha)	Sites >100 ha	Sites <10 ha	All Sites	Av. Area (ha)
England	32,500	59	16	120	270.8
Scotland	6748	14	280	380	17.8
Wales	6089	8	15	57	106.8
N. Ireland	239	0	6	15	15.9
Total	45,337	81	304	577	78.6

Table 8.1. Regional distribution of saltmarsh sites in the UK, showing distribution of sites by size (from Burd, 1995 and P. Corbett, pers comm.)

Site Name	County	Area (ha)
Tamar Estuary	Devon	64.74
River Avon	Devon	25.53
Dart Estuary	Devon	25.0
Erme Estuary	Devon	20.75
Teign Estuary	Devon	13.0
River Yealm Complex	Devon	2.46
Exe Estuary (Goosemoor)	Devon	6.0
		Total Area (ha)=157.48

Table 8.2. List of saltmarsh sites within the THESEUS Plymouth, UK (from Burd, 1989)

Site Name	County	Habitat type
Blaxton Meadows, Saltram, Plymouth	Devon	Saltmarsh
South Efford Marsh, Kingsbridge	Devon	Saltmarsh
Goosemoor (See link below)	Devon	Saltmarsh

Table 8.3. Saltmarsh creation sites through managed retreat within the THESEUS Plymouth, UK (from Atkinson *et al.*, 2001)

COASTAL DUNES

Geographical differences in dune vegetation are complex and several axes of variation exist: there is at least as much variation east-west as north-south. Community distribution reflects differences in species distribution. Sediment supply comes from eroding coastlines, particularly cliffs, and from the seabed through reworking of glacial deposits. Material is washed up on the shore and is redistributed by the wind when the surface dries out between tides. A breeze of just over 4.5m/s will start moving dry sand. The seeds of some species of colonising plants can germinate and grow on the strandline. Including *Atriplex* spp. and then, beyond the reach of all but the most extreme tide, *Honckenya peploides* or *Cakile maritima*. Around these strandline plants and detritus small patches of sand can accumulate, and are prone to invasion by *Elytrigia juncea, Leymus arenarius* or *Ammophila arenaria*, thereby initiating foredune formation. Embryonic dunes are transient and will either be replaced by marram-dominated vegetation or washed away by storms. A supply of new sand is vital for the continued existence of the embryonic community and the long-term survival of the dune ecosystem. Out of the reach of the highest tide, but where there is still active sand movement, the sandbinding Ammophila arenaria is usually dominant. This community is also maintained by constant change and will disappear if the sand stabilises. Where the dune stabilises and the organic content of the soil increases, fixed dune grassland develops. Accretion as a result of succession can lead to a rise in soil levels and the establishment of Salix repens, which may lead to scrub or maybe woodland. Dune slacks form by erosion of dunes down to the water table; wet sand is not moved by wind. In some areas where there is damaged plant cover, natural processes or wind erosion can lead to 'blow-outs'. These rarely exceed 30m in length before natural stabilising factors come into force. Cycles of erosion followed by stability are part of the natural development of dunes and are essential to the maintenance of diversity. If there is severe, localised erosion from anthropogenic activity, such as that adjacent to car parks, traditional management techniques could be considered, such as fencing off areas or, ideally, building a boardwalk through the dunes.

Zonation is a fundamental attribute of a dynamic sand-dune ecosystem. The range of vegetation zones and the transitions between them should be maintained. In most cases there will be several distinct sand dune zones, typically strandline (with *Cakile maritima*, *Honckenya peploides, Atriplex* spp.), embryonic dune (sparse cover of *Elytrigia juncea*, *Leymus arenarius*), mobile dune (more stable dune dominated by *Ammophila arenaria*) and fixed dune grassland (with grasses such as *Festuca rubra*, *Festuca ovina* and herbs such as *Galium verum*, *Rhinanthus minor*, *Galium saxatile*). The hindshore may have dune slacks (with *Hydrocotyle vulgaris* or *Salix repens*) or areas of dune heath (considered under separate guidance).

Figure 8.3. Sand dunes of Dawlish Warren have been subject to extensive management in order to protect low-lying land in the adjacent Exe Estuary

BIOGENIC REEFS SEA GRASSES

Sabellaria alveolata and Sabellaria spinulosa

In Britain, *Sabellaria* reefs are found only on shores with strong to moderate wave action in the south and west, between Lyme Bay on the south coast of England and the Scottish coast of the Solway Firth. The reefs have also been found on parts of the Northern Ireland coast. The British Isles represent the northern extremity of the range in the north-east Atlantic, which extends south to Morocco. The reefs also occur in the Mediterranean. Individual worms have a lifespan of typically three to five years, and possibly up to nine years, but reefs themselves may last longer as a result of further settlement of worms onto existing colonies. Typically in the first two years or so, after heavy settlement, there are few associated species. Over time, seaweeds including fucoids, *Palmaria palmata*, *Polysiphonia* spp., *Ceramium* spp., *Enteromorpha* spp. and *Ulva lactuca*, and animals including barnacles, dogwhelks, winkles, mussels and other bivalves such as *Nucula nucleus*, *Sphenia binghami* and *Musculus discors*, colonise the reef. Small polychaetes such as *Fabricia stellaris*, *Golfingia* spp. and syllidae predators may occur within the colonies. Blennies, small crabs (*Carcinus maenas*) and other crustacea (such as *Unicola crenatipalma*) can be found within crevices. Older reefs may increase the biodiversity and stability of what would otherwise be sand abraded rocks and boulders. Sheet-like reefs may restrict drainage of the shore, creating rockpools where there would otherwise be none. Less is known about subtidal communities.

In Britain, *Sabellaria* forms well developed reefs over much of its range. The most numerous and extensive areas occur on the Cumbrian coast, particularly between the Morecambe Bay and the Solway Estuary and at Dubmill Point. Reefs are also found in Cardigan Bay and in the Bristol Channel, including the coasts of south Wales, north Devon, Somerset and Avon. Very extensive subtidal reefs occur in the Severn Estuary, and subtidal populations have also been reported in the Walney Channel (Morecambe Bay) and from Glassdrumman, Northern Ireland. There is evidence of a significant contraction in range on the south coast of England over a period of at least 20 years until 1984. Declines have also been reported in the western part of the north Cornish coast, the upper parts of the Bristol Channel and in North Wales and the Dee Estuary. Causes have not been postulated and it is difficult to assess the true significance of these changes given the natural variability of the species.

Table 3: General biology of *Sabellaria alveolata*

Typical Abundance	High density		
Male size range	3 -4 cm	**Male size at maturity**	Field unresearched
Female size range	3 -4 cm	**Female size at maturity**	Field unresearched
Growth Form	Tubicolous Vermiform segmented	**Growth rate**	12 cm/year
Body Flexibility		**Mobility**	Permanent attachment
Characteristic feeding method	Active suspension feeder	**Typically feeds on**	Seston
Sociability	Gregarious	**Environmental position**	Epifaunal

Sabellaria alveolata distribution

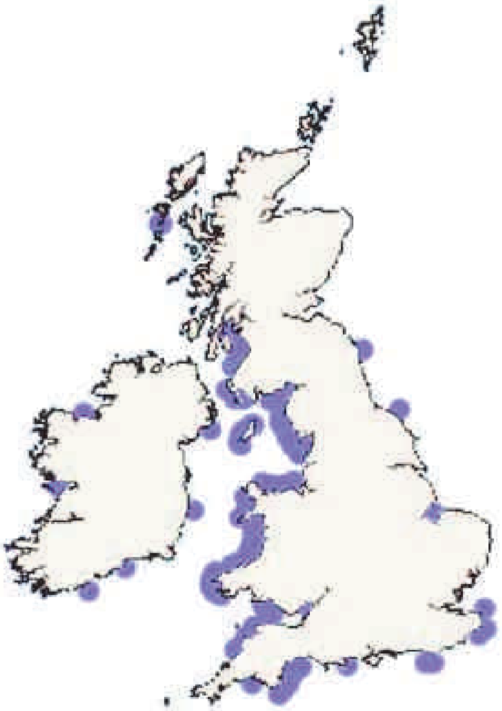
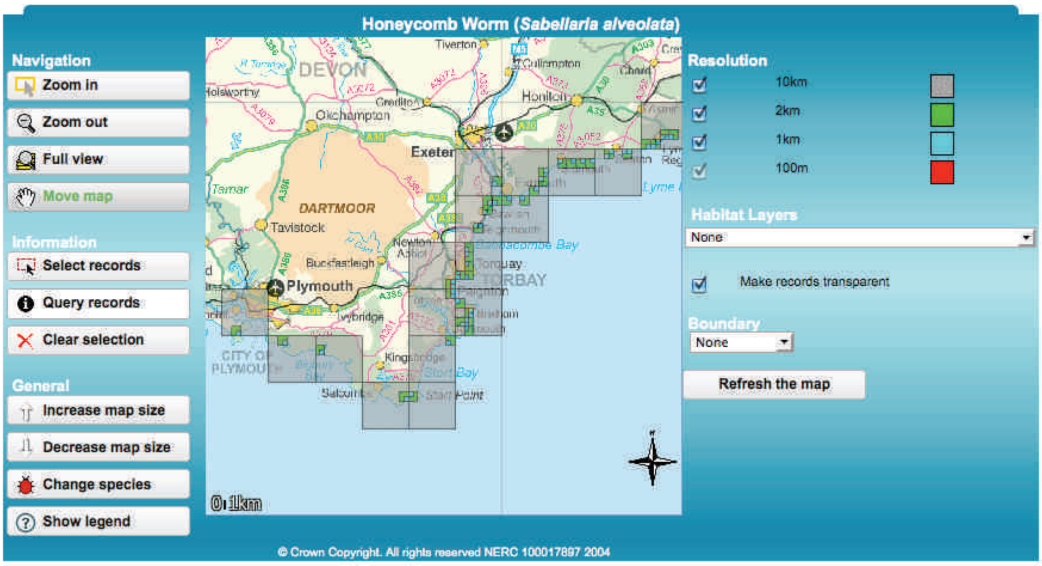


Figure 8.4. Sabellaria alveolata recorded (●) and expected (●) distribution in Britain and Ireland

Figure 8.5. Sabellaria alveolata

In Britain, most abundant on the south and west coasts with isolated records form the south east and east coasts. The northern limit is the Outer Hebrides. It is also found on south, west and north coasts of Ireland (q.v. Figure).



Resolution
☑ 10km ▨
☑ 2km 🟩
☑ 1km ☐
☑ 100m 🟥

Habitat Layers: None

☑ Make records transparent

Boundary: None

Figure 8.6. Distribution of *Sabellaria alveolata*

Sabellaria spinulosa

An occasionally gregarious segmented worm that builds tubes from sand or shell fragments. Found subtidally in exposed areas. Does not form reefs over most of its range being found mostly individually but may form thin crusts or large reefs up to several metres across and 60 cm high (q.v. the Marine Life Information Network website).

Typical Abundance	High density		
Male size range	20 - 30 mm	**Male size at maturity**	Field unresearched
Female size range	20 - 30 mm	**Female size at maturity**	Field unresearched
Growth Form	Vermiform segmented Tubicolous	**Growth rate**	Field unresearched
Body Flexibility		**Mobility**	Permanent attachment
Characteristic feeding method	Active suspension feeder	**Typically feeds on**	Phytoplankton
Sociability	Solitary Gregarious	**Environmental position**	Epifaunal

Table 8.4. General Biology Sabellaria spinulosa

Sabellaria spinulosa distribution all British and Irish coasts

Figure 8.7. Distribution os *Sabellaria spinulosa* in Britain and Ireland

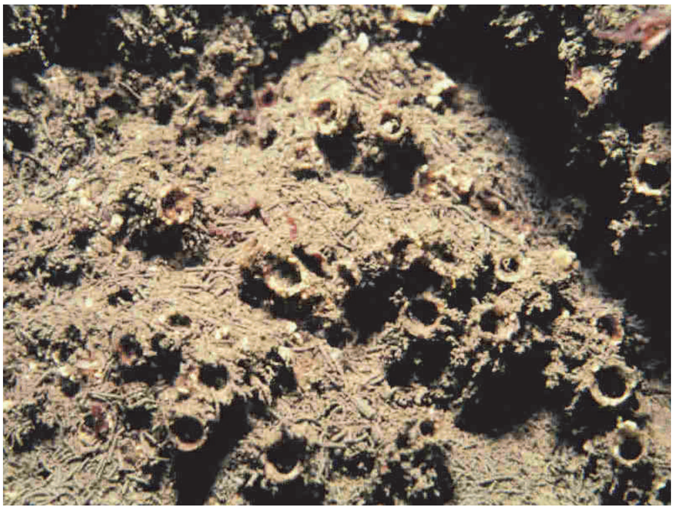
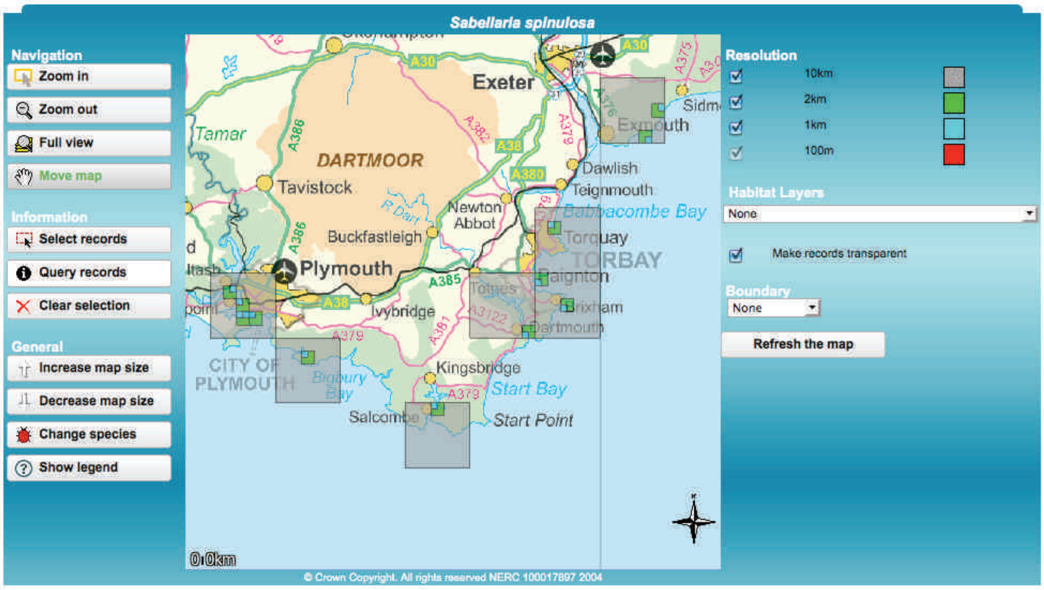


Figure 8.8. *Sabellaria spinulosa* reef



Resolution
☑ 10km ▨
☑ 2km 🟩
☑ 1km ☐
☑ 100m 🟥

Habitat Layers
None

☑ Make records transparent

Boundary
None

Figure 8.9. Distribution of *Sabellaria spinulosa*

Mussel bed

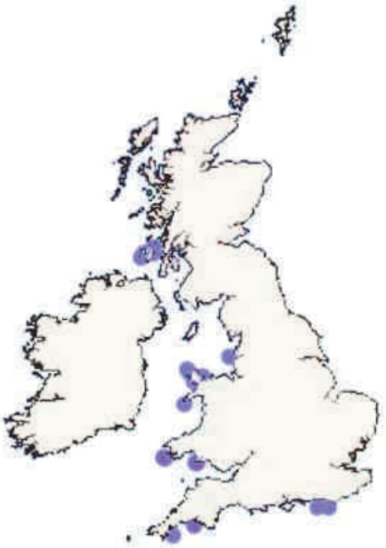
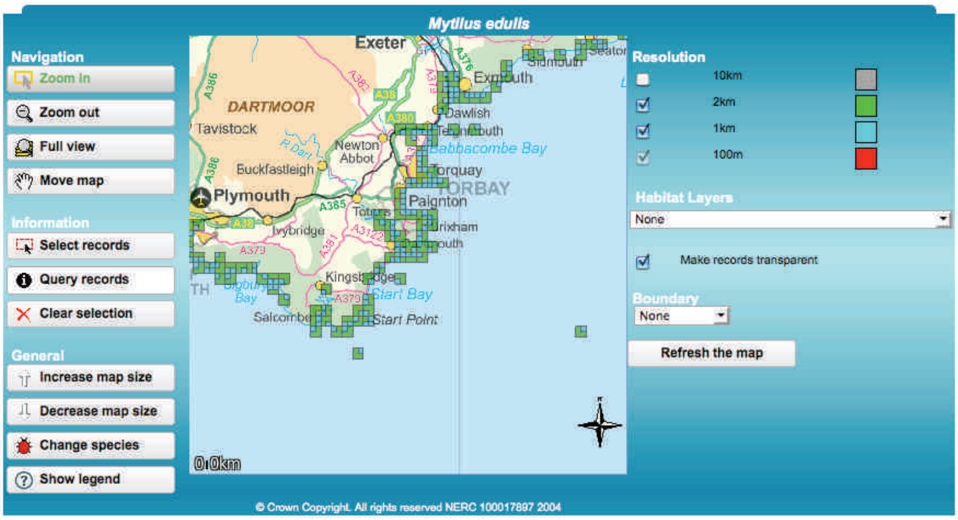


Figure 8.10. Distribution of the *Mytilus edulis* in UK:



Resolution
☐ 10km
☑ 2km
☑ 1km
☑ 100m

Habitat Layers: None

☑ Make records transparent

Boundary: None

Figure 8.11. Distribution of *Mytilus edulis*

Description of management status

Sabellaria spinulosa

There is currently no statutory protection for known examples of this sublittoral habitat in the UK. The marine SACs list is incomplete with respect to biogenic reefs, although Sabellaria spinulosa reefs may represent important sub-features of other Annex I habitats for which a site was selected. None of the candidate Special Areas of Conservation (cSACs) were selected specifically for biogenic reefs, although they may represent important sub-features of other Annex 1 habitats ('Sandbanks which are covered by seawater at all times'; 'Large shallow bays and inlets', and 'Estuaries'), for which a site was selected. Environmental assessments carried out prior to aggregate extraction operations could result in refusal of licences by the Department of the Environment, Transport and the Regions (DETR), or imposition of conditions to minimise dredging impacts.

Sabellaria alveolata

Intertidal protection for *S. alveolata* reefs can be achieved through SSSI designation. S. alveolata reefs also occur as sub-features of non-reef Annex 1 habitats (eg intertidal mudflats and sandflats) under the Habitats Directive and are present in a number of candidate Special Areas of Conservation (cSACs).

Mytilus sp.

Though *Mytilus* spp. is not designated under any protection laws, the habitat "Intertidal *Mytilus* spp. beds on mixed and sandy sediments" has been listed on the OSPAR List of Threatened &/or Declining Species and Habitats. *Mytilus* spp. is also protected by fisheries regulations. Fisheries regulations vary greatly in different parts of the Europe. The regulatory considerations in terms of mussel fisheries management are complex.

Current methods for management

Biodiversity is of intense interest for managers and policy-makers. As such, The United Nations declared 2010 the International Year of Biodiversity (Resolution 61/203). Throughout the course of the year events will take place world-wide to raise public awareness of not only the biological diversity on our planet, but the importance of protecting it. The origins of legal mechanisms and targets for protecting biodiversity mostly stem from the Convention on Biological Diversity (CBD) that was drawn up in 1992. Parallel to the CBD, the European Community (EC) adopted the Council Directive 92/43/EEC in 1992, this legalisation became more commonly known as the Habitats Directive. The directive focussed on the conservation of natural habitats and of wild fauna and flora through the establishment of a network of Special Areas of Conservation (SACs). The primary objective of which, is to promote the safeguarding and preservation of threatened species and habitats deemed to be of European importance.

In response to the CBD, the UK Government also founded the UK Biodiversity Partnership to develop and UK Biodiversity Action Plans (UK BAP). UK BAP recognises threatened biological assets within the UK and its surrounding waters and presents policies for the management and conservation of these assets. Plans for species and habitats in danger have been established to aid in recovery in order to assist in the UK"s development in reducing biodiversity loss set out in the CBD (UK Biodiversity Group, 1999). To date it has lead to the construction of action plans for 1150 priority species and 65 priority habitats (BRIG, 2007). Reefs are one of the habitats listed under Annex I of the Habitats Directive which require the designation of an SAC.

Intertidal Mudflats, Saltmarsh & Seagrass (*Zostera*)

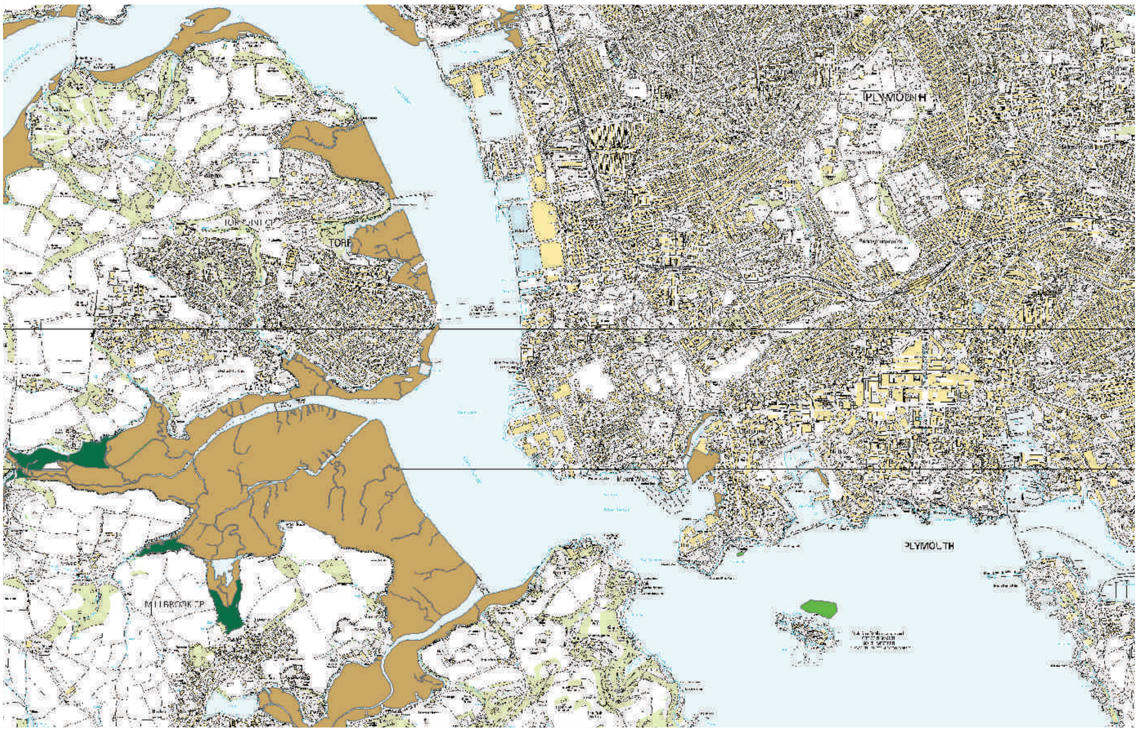


Figure 8.12. Map of the natural habitats in the Plymouth estuary (light green: Zoostera sp; dark green: salt meadows; brow: mud)

Seagrass beds help to stabilise shorelines, by reducing the impact of waves and currents and in addition by providing an extremely important habitat for fish and wildlife. Seagrass beds typically occur as broad, shallow meadows under low velocity regimes and their influence on wave energy can be substantial (Fonseca and Cahalan 1992). Seagrasses can provide a two fold benefit in coastal protection by attenuating wave energy and underpinning sediment and beach stability. This is achieved by slowing water motion and current flow and by reducing sediment suspension, whilst directly influencing sediment composition through accretion of biological particles from the seagrass itself and by retaining fragments of shell and skeletons of the numerous organism that inhabit the meadow (Fonseca and Cahalan 1992; Borum et al 2004). Frequently forming extensive monospecific beds, seagrasses are considered fragile and vulnerable habitats because of their physical morphology and growth patterns, and their position in coastal waters subject to frequent environmental perturbation (Fortes, 1988). Both natural and anthropogenic disturbances are responsible for the loss and fragmentation of seagrass habitats (Frost et al. 1999).

Seagrasses in South West England

In the UK seagrass beds in the Southwest suffered a severe population decline in the 1930s as a result of disease. Since the 1930s, seagrass populations have only made a partial recovery and the disease is still thought to be present (Saunders et al., 2003). However seagrasses are still present in all their native habitats in the region, including saline (saltwater), estuaries, such as Salcombe and Exe, and sheltered bays, such as Torbay. Some sites in the Southwest are known to be important as a nursery area for fish and cephalopods, such as cuttlefish. Unfortunately, many sites consist of small fragmented populations making them vulnerable to total loss (Frost et al., 1999).

Seagrass beds have been recorded at several sites within the Plymouth study site. These include Plymouth Sound, the mouth of the Yealm Estuary, the west and east bank of the Salcombe Estuary, Torbay and the Exe Estuary (Figure 56).

Distribution

In Plymouth sound Zostera marina forms beds adjacent to Jennycliffe Bay, close to Drakes Island, and in Cawsand Bay (Sounders et al., 2003) (Figure 57). A survey carried out by Bugg and colleagues (2004) reports that the Jennycliffe bed is fairly sparse, though some patches are denser and generally have longer shoots. The patchy formation of this bed reflects high water motion in the area. In accord with Bugg (2004), the bed, grows on a sandy substrate and is not currently under threat by any non-native species, such as S.muticum, but does support hydroids and epiphytic growth. It does appear that the Jennycliffe bed could be affected by the wasting disease, as divers reported black spots on some of the shoots (Bugg 2004).

The bed at Asia Shoal, north of Drakes Island (50°21.5’N, 04°09.3’ W) has Zostera marina which thins to muddy sand and pebble on the edge of the deeper channel slope. In Cawsand Bay on the western side of Plymouth Sound (50°20.0’N, 04° 11.5’ W), sporadic seagrass was reported and the site has been identified as a priority for further investigation (Devon Wildlife Trust report, 1993). The bed at this site may also have been affected by the wasting disease in the early 1930's and by outfalls discharging into the bay.

An extensive bed of seagrass is present in Cellars Cove at the mouth of the Yealm estuary (50°18.5’N, 04°04.0’) (Figure 58). It is a shallow (6m depth) sublittoral bed of *Zostera marina* adjacent to unvegetated areas.

In the Salcombe-Kingsbridge Estuary, Devon, UK (50°13'N, 03°46'W) seagrass beds are located on both west and east bank (Tweedley et al., 2008) (Figures 59, 60). The Salcombe Estuary comprises a total area of 674 ha of which 446 ha is intertidal, with the lower estuary being characterized by rocks and sandy bays (Frost et al., 1999). The seagrass bed on the west bank is a continuous meadow of ca. 2.3 ha, whilst the bed on the east bank of the estuary is fragmented into patches of 6–9 m^2 (Frost et al., 1999).

Torbay contains a total area of over 80 hectares of Zostera marina, representing 30% of all of Devon’s Seagrass beds (Figure 61). At Torre Abbey Sands off Torbay (50°27’N, 03°58’W) the bed of Zostera marina L. is exposed at extreme low water (Hirst and Attrill, 2008). However coverage of the intertidal bed is sparse and is made up of small patches of seagrass ranging from a few shoots to patches up to 1.6 m across, surrounded by sand, with more contiguous coverage present in the subtidal (Hirst and Attrill, 2008)..

In the Exe estuary *Zostera marina* gives the way to extensive Zostera noltii beds with localized small patches of Zostera angustifolium. The Exe estuary is 10 km long and 1.2 km wide. Wigeon and Brent Goose feeding was studied at two Zostera beds, Cockwood Corner (≈ 25.1 ha, a protected nature reserve, 50°36’N 03°23’W) and Cockle Sands (an unprotected area of 101.5 ha subject to higher levels of disturbance, 50°37’N 03°24’W).

Figure 8.13. Distribution of Seagrass beds within the Plymouth study site

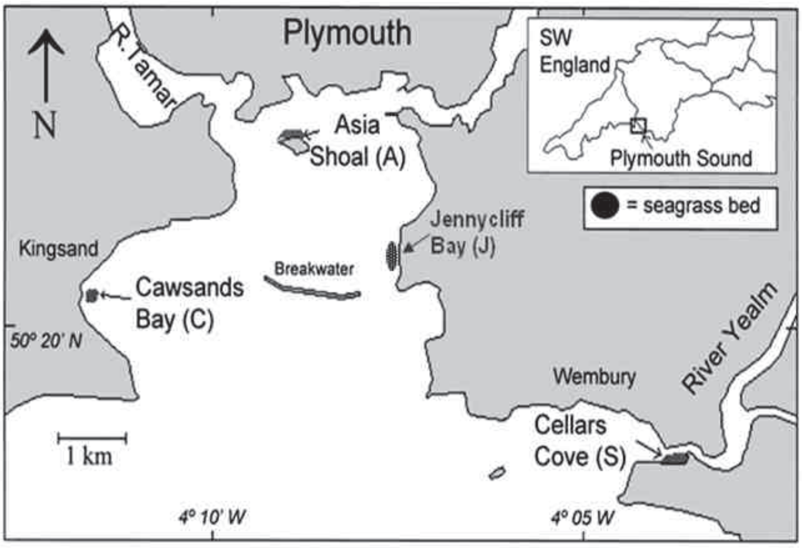


Figure 8.14. Seagrass beds within the Plymouth sound and the Yealm estuary. Figure modified from Saunders et al., 2003

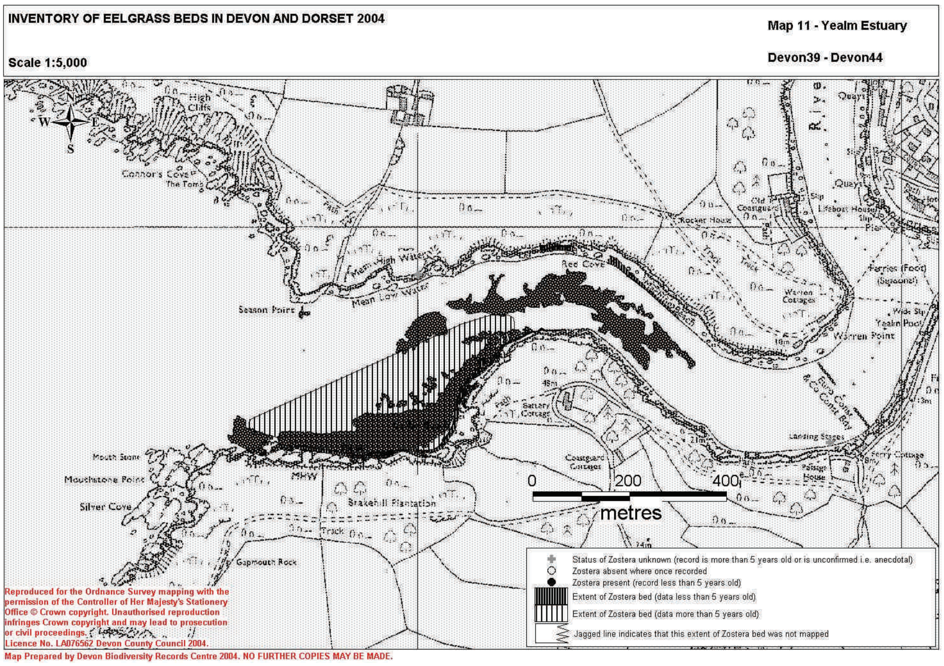


Figure 8.15. Seagrass in Cellar cove (Yealm estuary) (courtesy of Nigel Mortimer give his affiliation)

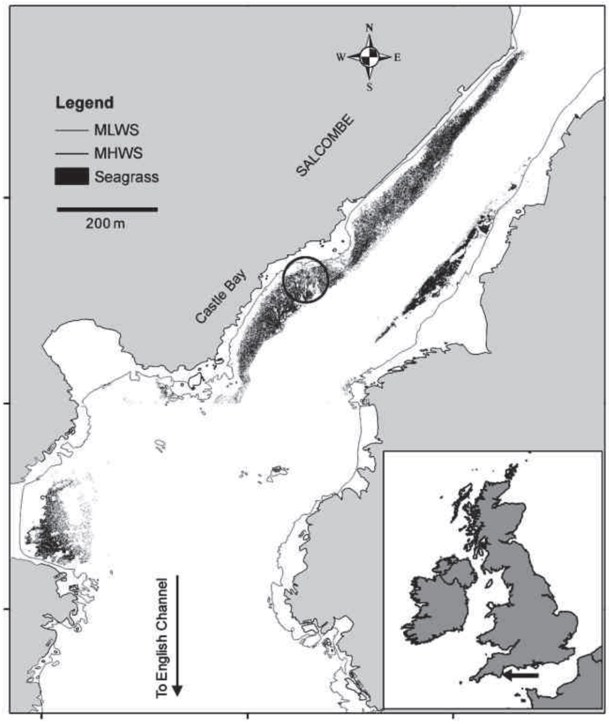


Figure 8.16. Seagrass beds distribution within the Salcombe estuary. Circle indicates the sampling location of the study conducted by Tweedley et al., 2008. The shaded area indicates the location of seagrass beds. MLWS: mean low water springs; MHWS: mean high water springs

Figure 8.17. Seagrass bed in Mill Bay, Salcombe Estuary. Photo: Emma Jackson

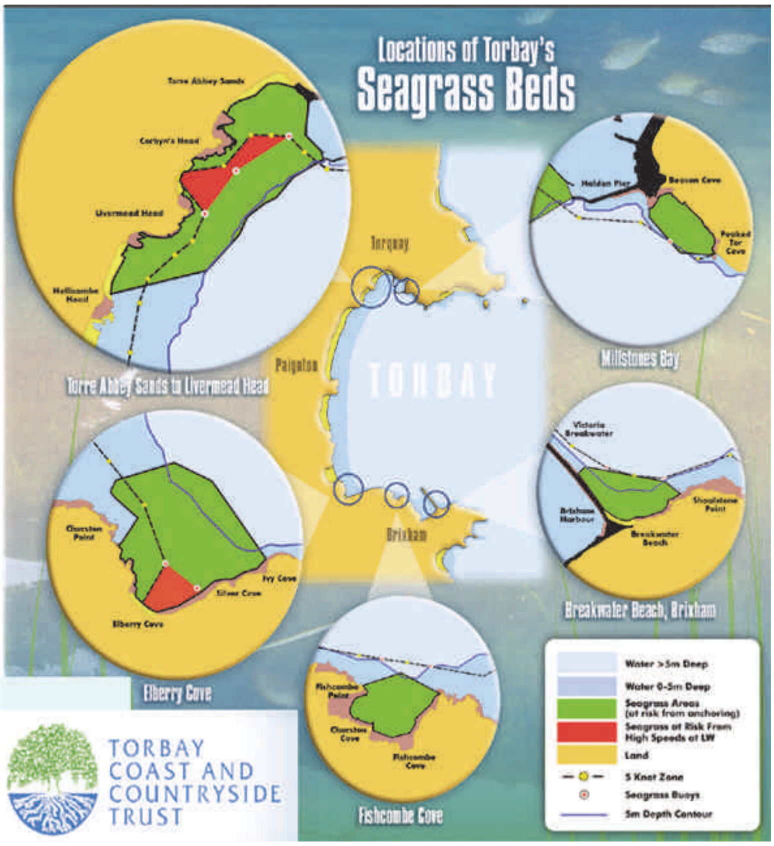


Figure 8.18. Distributon of seagrass beds in Torbay (courtesy of Torbay Coast and Countrside Trust)

North Wales overview

Biogenic reefs present in North wales are described in the following section, using data provided by universities of Wales Bangor.

Maps (Figure to Figure) were taken from the NBN Gateway website on November 18th 2010. Green squares represent distribution at 2 km scale; blue squares represent distribution at 1 km scale. Information on changes in patterns of distribution over time is also presented (Table).

Modiolus modiolus beds

Modiolus modiolus horse mussel beds EUNIS Code: A5.621, A5.622, A5.623 and A5.624. *Modiolus modiolus* forms dense beds, at depths up to 70m mostly in fully saline conditions.

Although *M. modiolus* is a widespread and common species, horse mussel beds (with typically 30% cover or more) are more limited in their distribution. *Modiolus* beds are found on a range of substrata, from cobbles through to muddy gravels and sands, where they tend to have a stabilising effect.

Maps of the distribution of *Mytilus edulis* and *Modiolus modiolus* in North Wales

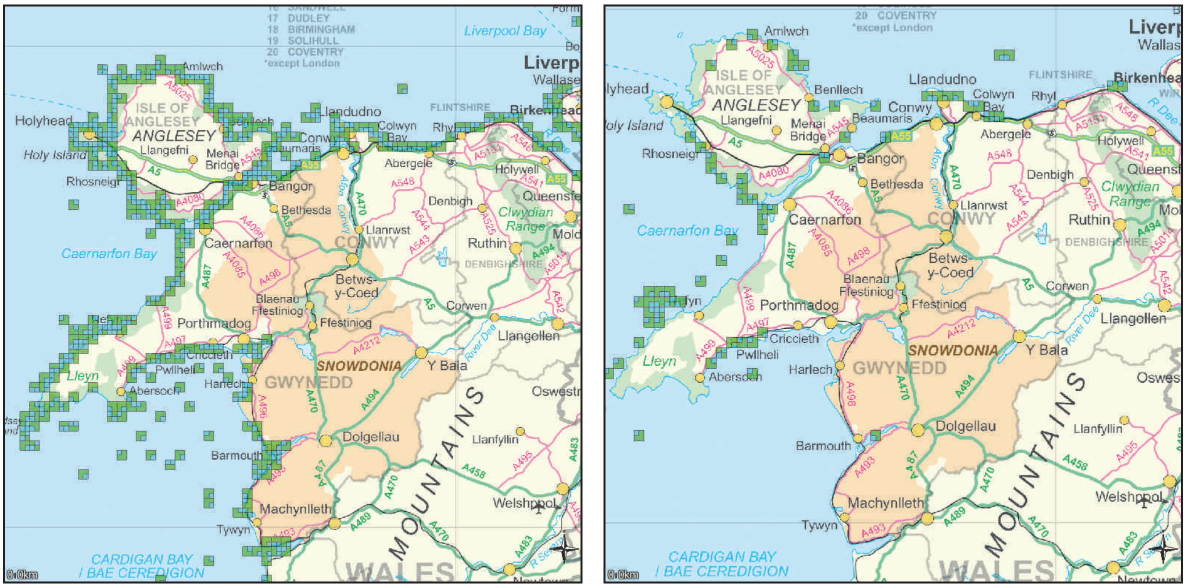


Figure 8.19. Distribution in North Wales of *Modiolus modiolus* (left) and *Mytilus edulis* (right)

Map of the distribution of *Ostrea edulis* in North Wales

Figure 8.20. *Ostrea edulis* distribution in North Wales.

Maps of the distribution of *Sabellaria alveolata* in North Wales

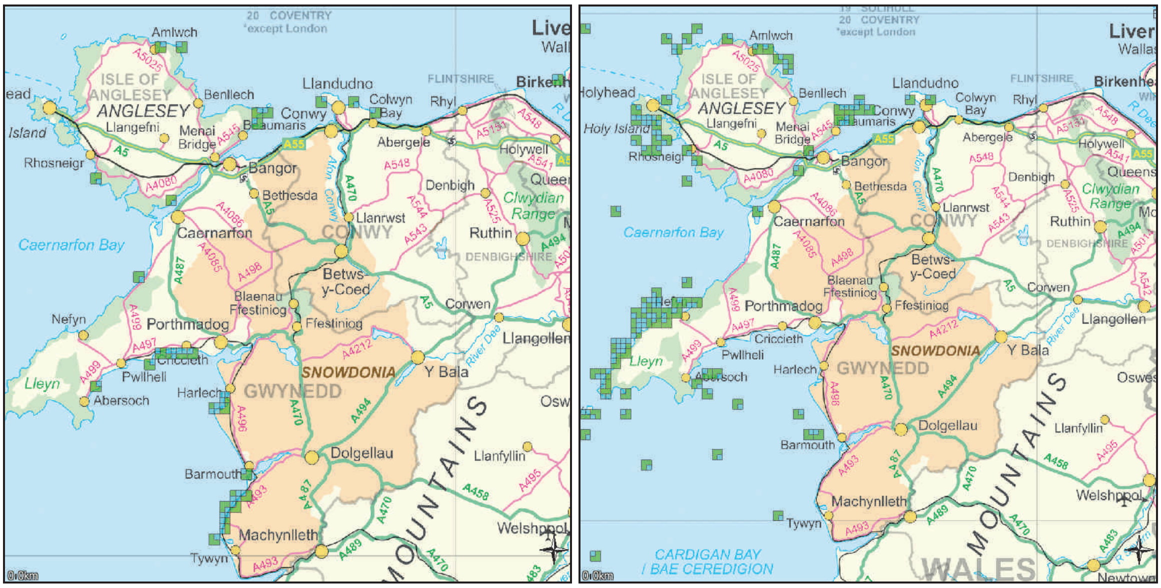


Figure 8.21. Distribution in North Wales of *Sabellaria spinulosa* (left) and *Sabellaria alveolata* (right)

Map of the distribution of *Ruppia maritima* in North Wales

Figure 8.22. *Ruppia maritima* distribution in North Wales

Table 8.5. Database of temporal variability in the abundance of *Sabellaria alveolata*

Historic data (pre 1984 - Cunningham et al 1984 report & www.nbn.org)			
Location	**Lat**	**Long**	**Date**
Penrhyn Bay	53,31547	3,74045	1923
N Llwyngwril Beach (Barmouth)	52,6762	4,078808	1982
Colwyn Bay	53,29756	3,720975	1950
Hilbre	53,39461	3,202006	1979
Hilbre	53,39461	3,202006	1919
Hilbre	53,39461	3,202006	1940
Barmouth	52,71849	4,052864	1978
Borth	52,49575	4,052736	1935
Shell Island	52,81882	4,147944	1978
Llandanwg	52,84359	4,120953	1978
Criccieth	52,91413	4,244878	1960
Criccieth	52,91413	4,244878	1954
Afen Wen	52,90819	4,320417	1964

Historic data (1984 only - Cunningham et al 1984 report)

Location	Lat	Long	Date
Tremadoc Bay	52,83994	4,123744	1984
Afen Wen	52,90819	4,320417	1984
Borth	52,49575	4,052736	1984
Tonfanau	52,62774	4,123794	1984
Barmouth	52,71849	4,052864	1984
Shell Island	52,81882	4,147944	1984
Llandanwg	52,84359	4,120953	1984
Criccieth	52,91413	4,244878	1984

Contemporary data (1996-date)

Location	Lat	Long	Date
Aberdaron	52,80166	4,721256	2004
Criccieth	52,91413	4,244878	2004
Llandanwg	52,84359	4,120953	2004
Shell Island	52,81882	4,147944	2004
Friog	4,147944	4,060058	2004
Cae-Du	52,63232	4,119583	2004
Tonfanau	52,62774	4,123794	2004
Porth Fechan	52,89729	4,325783	2005
Amlwch Discharge East - before discharge	53,41773	4,333031	2003
Penmon Priory to Black Rock (Menai Strait)	53,296	4,059275	1996/1997
Pwll Fanogl to Tal-y-Foel (Menai Strait)	53,15617	4,282919	1996/1997
W of Criccieth (Tremadog)	53,90944	4,255044	1996/1997
W Great Ormes Head (Llandudno)	53,33406	4,877875	1996/1997
Pen-ychain (Pwllheli)	53,89639	4,325733	1996/1997
SW of Aberdesach (Trefor)	53,03239	4,359894	1996/1997
S Puffin Island (N Menai Strait)	53,31533	4,033183	1996/1997
Porth Cwyfan (Rhosneigr)	53,18436	4,490972	1996/1997
Llangelynin beach (Barmouth)	52,63594	4,118281	1996/1997
Shell Island (Harlech)	52,81512	4,153697	1996/1997
N Llwyngwril Beach (Barmouth)	52,6762	4,078808	1996/1997
W Penmon Point (N Menai Strait)	53,31243	4,045056	1996/1997
Llanbedrog (rocky shore) (Pwllheli)	52,85375	4,477931	1996/1997
Criccieth Castle (E) (Criccieth)	52,91714	4,228669	1996/1997
E Lynas Point (Dulas Bay)	53,4133	4,283131	1996/1997
E Little Orme (Llandudno)	53,32666	3,773939	1996/1997
Llandulas	53,29425	3,634917	2010
Perch Rock, Wirral	53,44488	3,043978	2010
Hilbre Island, Wirral	53,39461	3,202006	2010
Tywyn	52,57799	4,097428	2010

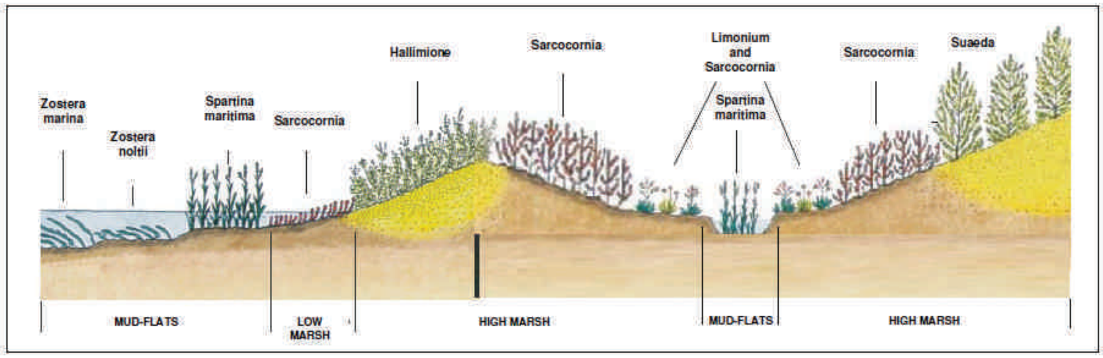
VIII.1.3 SANTANDER

SALT MARSHES

Figure 8.23. Aerial photograph of Santander Bay showing the distribution of salt marshes

Salt marshes description

HALOPHYTE MARSH (A):



SUBHALOPHYTE MARSH (B):

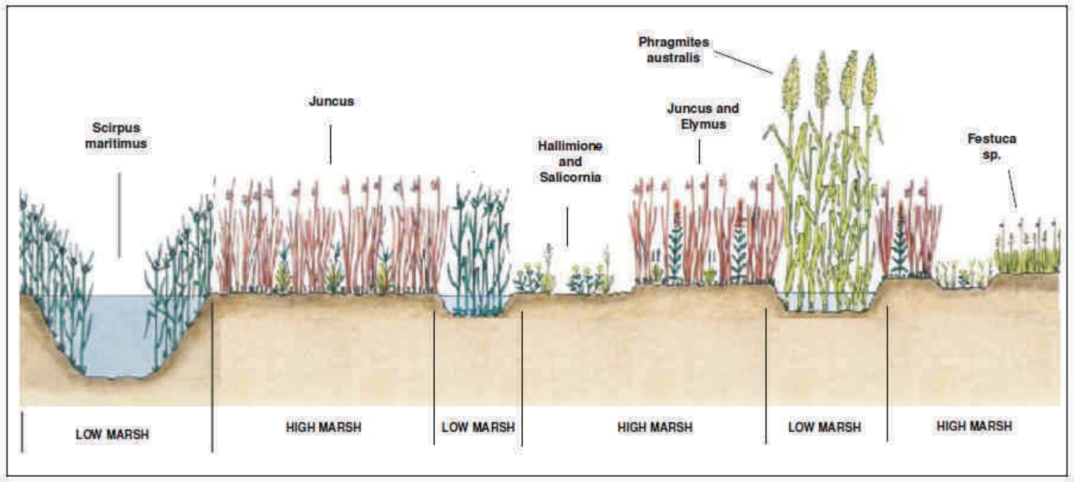


Figure 8.24. Succesion of vegetation in a halophyte mash (A) and a subhalophyte marsh (B)

Marsh management status

Marshes included in the Site of Community Importance (Natura 2000) ES1300005 Dunas del Puntal y Estuario del Miera, are managed by the regional authority of Biodiversity (Government of Cantabria). Marshes not included in the SCI are managed by the regional authority of Hydraulic works and water cycle (Government of Cantabria).

DUNES

Source : Universidad de Cantabria EID Méditerranée - 20/02/2011

Figure 8.25. Aerial photograph of Santander showing the distribution of dunes and adjacent vegetation types

Dominant species in sand dunes

Fourteen dominant vegetal species have been identified among the sand dunes placed in the Bay of Santander (Table).

Beach	Embryonic Dune	White Dune	Grey Dune
Cakile maritima	*Elymus farctus*	*Ammophila arenaria*	*Lagurus ovatus*
Salsola kali	*Eryngium maritimum*	*Carex arenaria*	*Plantago maritime*
	Honkenia peploides	*Euphorbia paralias*	*Crithmun maritimum*
		Eryngium maritimum	*Helycrisum stoechas*
		Calystegia soldanella	*Crucianella maritime*

Table 8.6. Database list of dominant species in Santander sand dunes

Dune management status

The dune system is included in the Site of Community Importance (Natura 2000) ES1300005 Dunas del Puntal y Estuario del Miera. The site is managed by the regional authority of Biodiversity (Government of Cantabria).

BIOGENIC REEFS

Mussel beds, Oyster beds and Sabellaria reef are not present in the Bay of Santander.

SEAGRASSES

Both *Zostera marina* and *Zostera noltii* are present at Santander and are known to provide doverse habitats for a range of marine fauna (Table and Table).

Figure 8.26. Aerial photograph of Santander Bay showing the distribution of seagrass

Dominant Species

Species	Frequency (%)	Species	Abundance (indiv/m²)
Abra alba	100	*Rissoa parva*	21768
Bittium reticulatum	100	*Hydrobia ulvae*	17132
Hinia reticulata	100	*Bittium reticulatum*	11652
Notomastus latericius	100	*Loripes lacteus*	5380
Rissoa parva	100	*Euclymene oerstedii*	5020
Euclymene oerstedii	94	*Abra alba*	4872
Loripes lacteus	94	*Notomastus latericius*	2854
Marphysa belli	94	*Hinia reticulata*	2780
Venerupis sp	87	*Marphysa belli*	1264
Owenia fusiformis	81	*Cyathura carinata*	1080
Carcinus maenas	75	*Lucinoma borealis*	1060
Lucinoma borealis	75	*Venerupis sp*	921
Lumbrinereis sp	75	*Euclymene sp*	880
Parvicardium sp	75	*Owenia fusiformis*	740
Hydrobia ulvae	69	*Perinereis cultrifera*	464
Phyllodoce sp	69	*Parvicardium sp*	416
Platynereis dumerilii	69	*Cerastoderma edule*	356

Table 8.7. Database list of dominant species in *Zostera* marina beds in the Santander Bay

Species	Frequency (%)	Species	Abundance (indiv/m²)
Hinia reticulata	100	*Hydrobia ulvae*	39004
Abra alba	92	*Bittium reticulatum*	10516
Hydrobia ulvae	92	*Hinia reticulata*	1568
Venerupis sp	92	*Rissoa parva*	1332
Bittium reticulatum	83	*Abra alba*	1036
Cerastoderma edule	83	*Notomastus latericius*	1036
Loripes lacteus	83	*Euclymene oerstedii*	912
Rissoa parva	83	*Parvicardium sp*	848
Cyathura carinata	75	*Cerastoderma edule*	788
Idotea chelipes	75	*Polydora sp*	732
Carcinus maenas	67	*Loripes lacteus*	628
Nephtys hombergii	67	*Venerupis sp*	556
Notomastus latericius	67	*Odostomia sp*	524
Odostomia sp	67	*Cyathura carinata*	420
Owenia fusiformis	67	*Idotea chelipes*	304

Table 8.8. Database list of dominant species in *Zostera noltii* beds in the Santander Bay

Seagrasses management status

Zostera marina and *Zostera noltii* meadows are included in the Site of Community Importance (Natura 2000) ES1300005 Dunas del Puntal y Estuario del Miera. The site is managed by the regional authority of Biodiversity (Government of Cantabria).

Coastal habitats variability

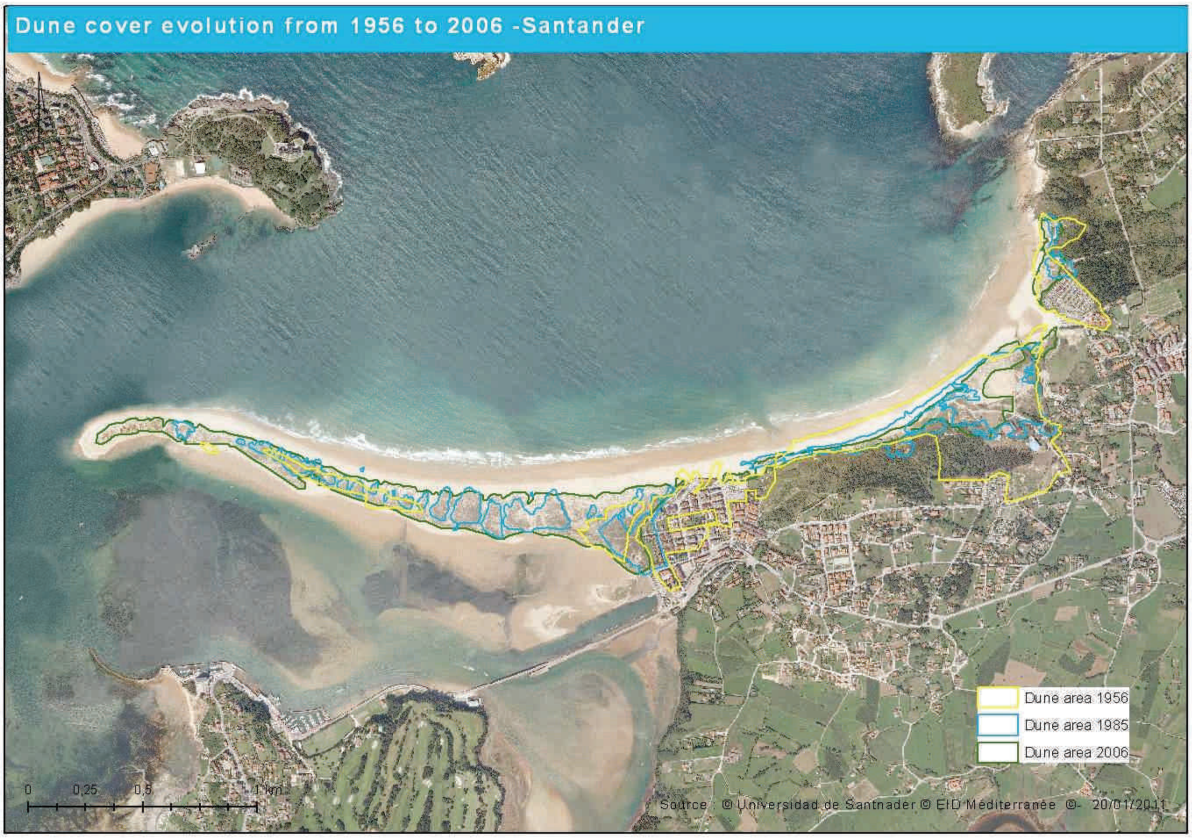


Figure 8.27. Aerial photograph showing variability in the extent of sand dunes in the Santander site 1956- 2006

A general retreat in extent is observed on the east part of the dune system, while a progressive extension is occurring on the western part of the system (Figure).

VIII.1.4 SCHELDT ESTUARY

General description

A typical estuary in which economical and ecological interest are at stake is the Scheldt-estuary. This estuary is part of the Scheldt basin and runs into the North Sea in the South-West of the Netherlands (Figure). Running through three countries the Scheldt catchment area is 22,116 km^2. The length of the river, from its origin in France to the mouth at Vlissingen (Flushing) is 355 km (Baeyens, et al., 1998). The last 160 km trajectory from Gent downstream is referred to as the Scheldt estuary. Tidal forces are strong along this part of the river, and fresh river water and salty sea water are responsible for a salt gradient. Because of this gradient and the river's extent, the Scheldt is unique and the last real estuary in Western Europe (Baeyens, et al., 1998; Meire, et al., 2005). Downstream of the Dutch-Belgian border to the mouth the estuary is called Westerschelde. The Westerschelde estuary is more diverse compared to the river, as in this part of the estuary multiple deep and shallow gullies alternate with salt-marshes, mud- and sand-flats. All wetlands in this estuary are protected under the Ramsar convention and EU legislation, Natura-2000 (Baeyens, et al., 1998; Meire, et al., 2005).

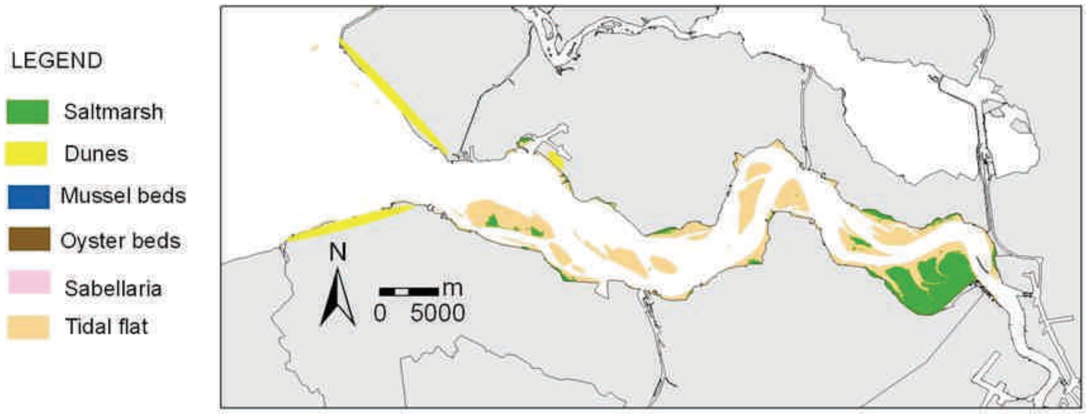


Figure 8.28. Coastal ecosystems along the Westerschelde

Next to its ecological value, the Scheldt is economically important because it links the harbours of Gent, Antwerpen, Terneuzen and Vlissingen to the North Sea. Consequently, keeping the navigation channels open is very important for local business and employment (SIC, 2009). A treaty between Belgium and the Netherlands has obliged the Dutch government to keep the Westerschelde open since 1839. Because of this agreement, maintenance dredging and dredge spoil deposition are executed continuously. In addition, capital dredging was carried out in the period 1970-1975 and 1997-1998 to deepen and widen the navigation channels to allow larger ships to reach Antwerp harbour, and to be less dependent on the tide. Further deepening is beign carried out at the moment (van der Wal, et al., 2008).

Through the centuries the estuarine landscape in the Scheldt basin has been reshaped to comply to satisfy human needs (Meire, et al., 2005). The scale of economic activity in the Scheldt basin has increased even more profoundly in the last century due to increasing human population and wealth. This has resulted in a significant increase in anthropogenic pressure on the estuary (Baeyens, et al., 1998; Meire, et al., 2005). Pollution, alteration and fixation of the navigation channels, and changes in land use have altered the initial natural landscape elements and biodiversity. These have either been severely degraded or lost entirely. Especially, large parts of the salt-marsh, mud-flat and other wetland area have been claimed and transferred into agricultural land. For instance, about 6040 ha estuary is acclaimed since 1900 matching a decline of 56% between 1900 and 1990. With these changes the estuary lost much of its dynamic nature (Meire, et al., 2005).

Salt-marshes

Compared to their original extent, tidal salt-marshes are mainly present as marginal fringes along the Scheldt-estuary (apart from the 'Verdronken land van Saeftinghe'), but some salt-marsh vegetation is establishing on the mud-flats in the middle of the estuary (Figure). Further degradation of the wetlands in the estuary, salt-marshes as well as mud- and sand-flats, is undesirable. Additional deepening of the navigation channel, shipping traffic and relative sea level rise might cause further degradation of these habitats in the estuary (Kirwan & Temmerman, 2009; Temmerman, et al., 2004). Although protected under Ramsar and Natura 2000 legislation, there is substantial economic pressure to further deepen the navigation channels to allow larges ships to enter the harbour of Antwerp.

Due to the influence of the river on the estuary there is a salinity gradient which influences the vegetation composition of the salt-marshes along the Westerschelde. Near the mouth mainly salt-water species can be found. For instance, the pioneering species English cordgrass (*Spartina anglica*) and glassworth (*Salicornai sp.*) are the main primary colonizers from the mouth upstream to Bath. From there on bullrush (*Scirpus maritimae*) is common as pioneer colonizing the mud-flat.

Management of the salt-marshes of the Westerschelde is scattered over different governmental and non-governmental organizations. Some marshes, and all dikes adjacent to them, are managed by governmental organization Scheldestromen (waterschap). Furthermore, two non-governmental organizations are responsible for the other salt-marshes: Stichting het Zeewse Landschap and Natuur Monumenten. Most salt-marsh area is 'unmanaged' and not grazed by cattle, providing rest and forage ground for many protected bird species. However, small fenced parts of Zuidgors and Baarland have some sheep grazing on them. Het verdronken land van Saeftinghe also has areas cattle grazed bt cattle on the high marsh.

Tidal-flats

The non-vegetated areas in the intertidal are mainly mud- and sand-flats. The mud-flats contain diversity on benthic species, and are valuable foraging and resting grounds for many bird species, like waders. Furthermore, seals use these mudflats for resting. Shellfish and lugworms are the most important species in the mud-flat habitats, but in some parts diatoms are also important, stabilizing the mud-flat surface preventing fine sediments from eroding (Weerman, et al., 2010).

The mud-flats are generally not managed, however, there is active maintenance dredging of the navigation channels, which affects the mud flats. Moreover, there are ongoing experiments with deposition of dredging material close to and at mud-flat edges. These experiments are carried out by the water and infrastructure administration.

Dunes

Dunes, and adjacent beaches, are mainly found in the mouth of the estuary, west of Vlissingen and Breskens. Here, these coastal ecosystems are the most important defence against the sea, instead of dikes that can be found stream upward. Because of the Delta law, large sections of these dunes have been reconstructed and reinforced during the last five years, since they were assessed to be weak links in the coastal defence. Although these reinforcement works are nearly finished, there is still some finishing work carried out. Together with the reinforcements some additional fresh-water wetlands have been developed behind these reinforced dunes.

Dunes are managed by the administration Scheldestromen (waterschap), who is responsible for all inland-waterbodies and dikes.

Seagrasses

Seagrasses play no important role in the Westerschelde. Some small patches of seagrass can be found near the mouth of the estuary in Vlissingen (salt-marsh of Ritthem). Management and protection of those spots occupied by sea grasses fall under the same scheme as mud-flats and salt-marshes.

Biogenic reef habitats

Species that structure biogenic reef habitats, like mussels, oysters or Sabalerria, seem to play no role of importance in the Westerschelde anymore. Although at some mud-flats there are commercially exploited mussel beds. This in contrast to the nearby Oosterschelde where, for example oysters, occupy a substantial area of about 6.82% of the intertidal mudflat, and is still increasing (Smaal, et al., 2009). Nevertheless, following range expansions in other parts of the Netherlands, like the Oosterschelde (Smaal, et al., 2009) and the Wadden Sea (Smaal, et al., 2005), the Pacific Oyster (*Crassostrea gigas*) has colonized mud flats in the Westerschelde as well. Along the whole Westerschelde Pacific Oysters can be found attached to hard structures, like dikes, breakwaters, and groins, new oyster reefs are starting to form on mud-flats west of Hoofdplaat (Personal observation, van Belzen 2011). The oyster reefs colonizing the mud-flats also facilitate settlement of mussels, as evidenced by the high abundance. However, at this moment no good survey data exists on the distribution of these reef structuring bivalves.

Sabellaria sp. presence in the Westerschelde is not recorded.

VIII.1.5 GIRONDE ESTUARY

Within the Gironde Estuary N2000 several sites are found including:

- NATURA 2000 SITE FR 7200678 : « Dunes du littoral girondin de la Pointe de Grave au Cap Ferret »
- NATURA 2000 SITE FR 7200697 : « Boisements à Chênes verts des dunes du littoral girondin »
- NATURA 2000 SITE FR 7200703 : « Forêt de la Pointe de Grave »
- NATURA 2000 SITE FR 7212014: « Marais du Blayais »
- NATURA 2000 SITE FR 7200684 : « Marais de Braud et Saint Louis et Saint Ciers sur Gironde»
- NATURA 2000 FR5412011 Estuaire de la Gironde : Marais de la rive nord
- NATURA 2000 FR7200680 Marais du bas médoc
- NATURA 2000 FR7200677 Estuaire de la Gironde
- NATURA 2000 FR7210065 Marais du Nord Médoc

A map has been prepared from this to indicate the extent of saltmarshes area around the Estuary. In addition Spartina swards have been located on the northern part of the Estuary (Figure).

Salt Marshes

Local map

Figure 8.29. Mapping of natural terrestrial habitats in the Gironde Estuary

Salt marshes are located around the Gironde estuary, the habitat Atlantic salt meadows shows variations of dominant species: *Halimione portulacoides* or *Elymus pycnanthus* or *Juncus anceps/Juncus maritimus*.

Atlantic salt meadows (Glauco-Puccinellietalia maritimae) present two subtypes:

Low schorre with *Puccinellia maritima*

High schorre with *Elymus atherica*

Spartina swards also develop in some areas of the estuary.

Figure 8.30. Picture of the Spartina swards, Les Darses, Le Verdon/Mer

Figure 8.31. Phragmites autralis community, with the village of Le Verdon/Mer in the background

Wetlands with *Phragmites australis*, *Bolboschoenus maritimus* and *Scirpus maritimus* are also quite common (Figure 8.31). While in the upper area of the Gironde estuary, in "Les Darses Area", five rare vegetal species have been found: *Linaria thymifolia, Aetheorhiza bulbosa, Silene conica, Ophrys passionis* and *Anacamptis laxiflora*. Also many protected birds are found: *Egretta garzetta, Hippolais polyglotta, Merops apiaster, Riparia riparia, Actites hypoleucos, Numenius phaeopus, Larus ridibundus, Cisticola juncidis, Haematopus ostralegus, Calidris alpine* and *Tadorna tadorna*. (Grand Port Maritime de Bordeaux, 2009).

Management status

A natural marine park is planned for the Gironde estuary and the Charente Maritime coast above. (c f aires marines website)). Many programs of knowledge and protection of the marine and coastal meadows are under process.

BIOGENIC REEFS ANS SEAGRASSES

Local map

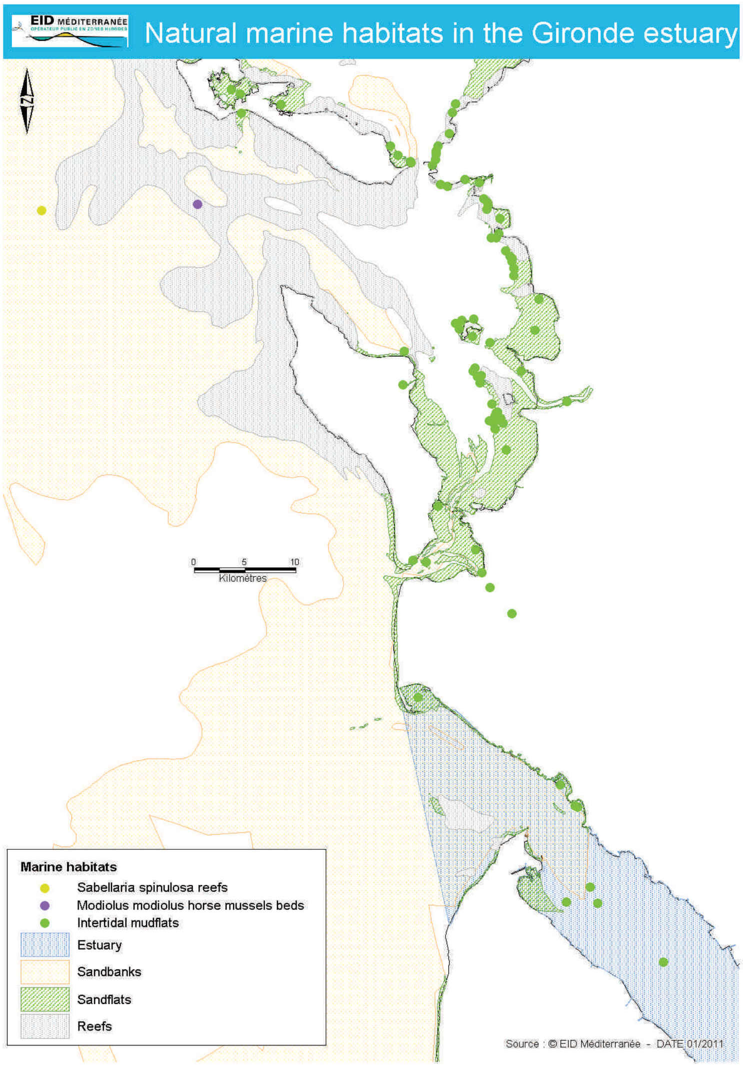


Figure 8.32. Natural marine habitats around Gironde estuary, data courtesy from AMP (Aires marines protégées) and OSPAR

The main marine habitats that can be found in the estuary are sandflats and Intertidal mudfltas. Biogenic reefs are located on the Charente maritime coast at the north of the estuary. Seagrasses are not present in the Gironde Estuary but we can find some meadows of Zostera marina and Zostera noltii more in the south on the Atlantic coast around Arcachon bassin.

Management status

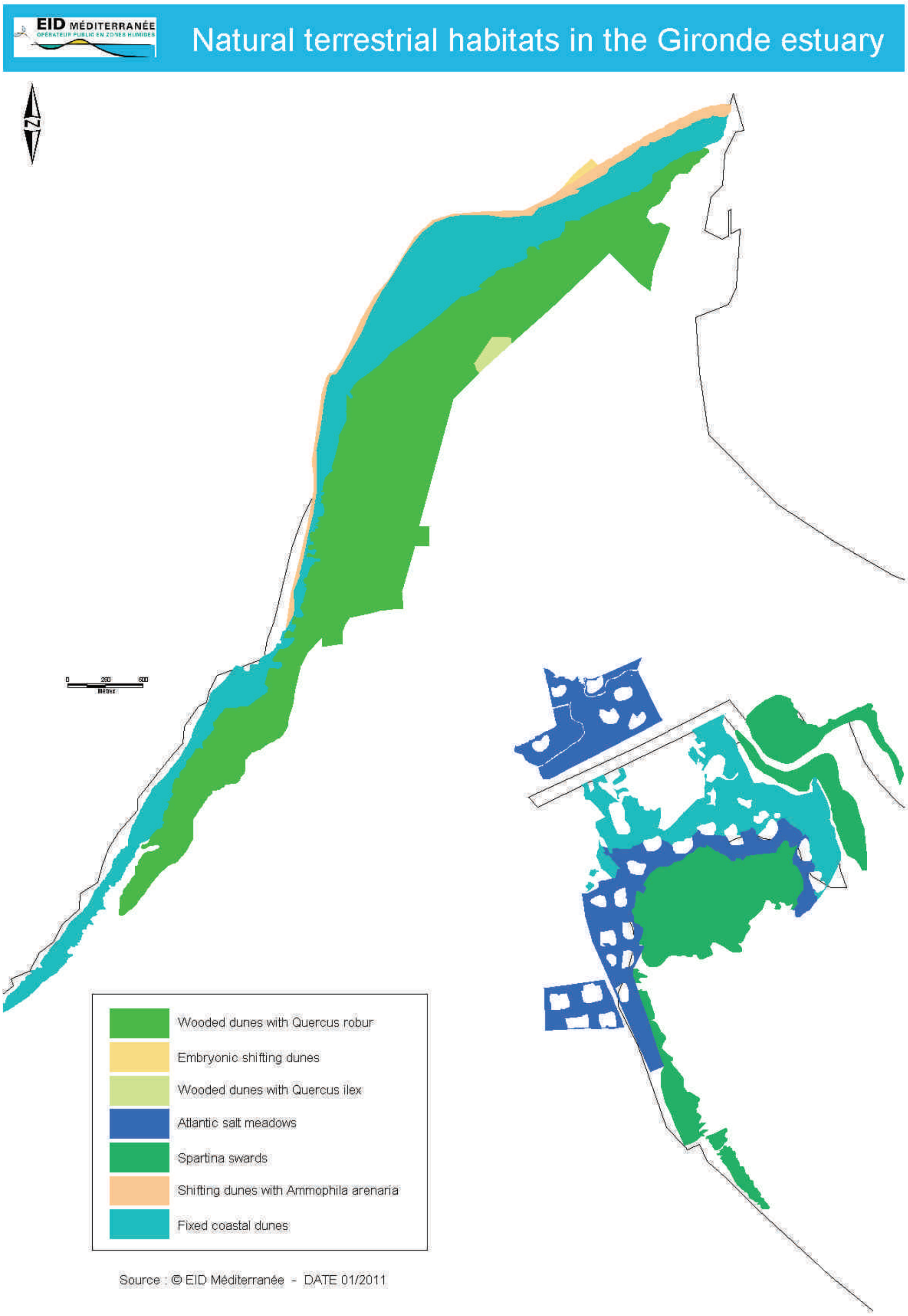
A natural marine park is planned for the Gironde estuary and the Charente Maritime coast above. (q.v. aires marines website). Many programs of knowledge and protection of the marine and coastal meadows are under process.

Coastal dunes

Local map

Source : © EID Méditerranée - DATE january 2011

Figure 8.33. Dunes habitats in the Gironde estuary



Source : © EID Méditerranée - DATE 01/2011

Figure 8.34. Natural terrestrial habitats in the Gironde estuary, focus on the Pointe de Grave

Presentation of dunes habitats

On the dunes of the Gironde pilot sites, the following habitats are present. The lists of vegetal species found by habitat and some pictures of each habitat are given below Figure to Figure :

Shifting dune

Ammophila arenaria	5
Euphorbia paralias	2
Eryngium maritimum	+
Calystegia soldanella	2
Medicago marina	+
Polygonum maritimum	+
Hieracium eriophorum	+

Figure 8.35. Picture of the shifting dune, Pointe de Grave, May 2010

Transitional fixed dune

Artemisia campestris	3
Hypochaeris radicata	1
Senecio vulgaris	1
Solidago virgaurea	+
Ononis repens	2
Myosotis sp	+
Aira caryophyllea	+
Helichrysum stoechas	2
Desmazeria marina	+
Phleum arenarium	2
Cerastium dubium	2
Aetheorhiza bulbosa	+
Cerastium diffusum	+
Corynephorus canescens	1

Figure 8.36. Picture of the transitional fixed dune, May 2010

Fixed dune

Lotus corniculatus	1
Artemisia campestris	3
Centaurea aspera	3
Solidago virgaurea	2
Cerastium dubium	1
Helichrysum stoechas	3
Pinus pinaster	+
Ononis sp	1
Ammophila arenaria	1
Lagurus ovatus	1
Euphorbia portlandica	+
Phleum arenarium	+
Festuca vasconcensis	1
Erodium cicutarium	+
Bryophyte	4

Figure 8.37. Picture of the fixed dune, Pointe de Grave, May 2010

Wooded dune

Cistus salvifolius	3
Quercus ilex	4
Quercus robur	2
Quercus pubescens	1
Arbutus unedo	1
Robinia pseudoacacia	1
Rubia peregrina	1
Ligustrum vulgare	+
Ruscus aculeatus	+
Pinus pinaster	1
Lonicera sp	+
Hedera helix	1

Figure 8.38. Wooded dune with *Quercus ilex* and *Quercus robur*

Management status

Coastal dunes in the Gironde estuary are managed by ONF for the and Conservatoire du littoral for the The embryonic, white, fixed and wooded dunes.

On the shifting dunes some restoration works were led using wooded fences acting as sand catchers and marasm grass plantings. The dunes are alos protected from trampling thanks to fences and organized pathways.

VIII.1.6 EMILIA ROMAGNA REGION (NORTH ADRIATIC SEA)

The Emilia Romagna coast is an area of renowned environmental, cultural and economical value, and one of the world's leading holiday destinations. In this rehion human pressure on the coast has been historically intense, leading to severe urbanization and overexploitation of natural resources, and developments have wiped out wetlands, dunes, shellfish beds and other valuable biogenic habitats. Environmental problems are varied and severe. Significant erosion and land subsidence (both natural and human-induced) have occurred since the beginning of the 20th century. This, together with poor coastal defence policies, has led to the proliferation of protection schemes, mainly groynes and offshore breakwaters. Nowadays over 60 % of Emilia Romagna shores are protected by defence structures (Figure), which has lead to enhancement of sheltered habitats, decrease of water quality, changes in the characteristics of sediments, loss of native sedimentary environments, and artificial enhancement of hard-bottom habitats (Bacchiocchi & Airoldi 2003, Bulleri & Airoldi 2007). Other major problems faced by coastal practitioners and managers are related to developments of settlements, industries, aquaculture, ports, military installations, tourism and recreation, which have lead to the creation of vast built-up areas at the expense of natural habitats (eg, dunes, saltmarshes).

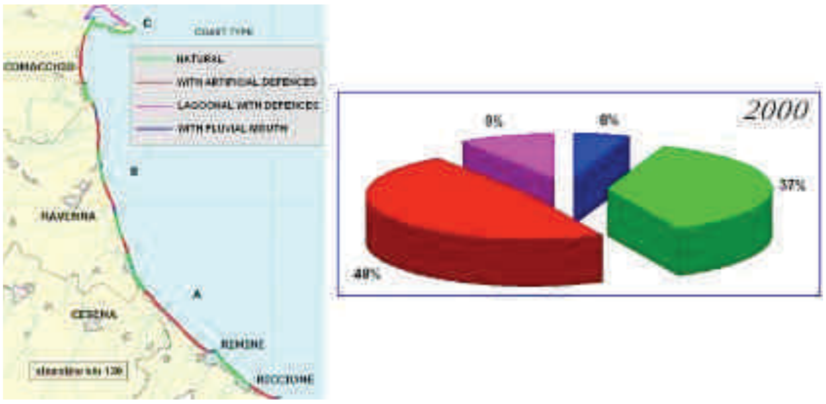


Figure 8.39. Coast-line types of Emilia-Romagna based on coastal protections and referred to year 2000 (source The Coastal Information System of the Region Emilia Romagna)

CESENATICO

Cesenatico offers a typical example of the conditions illustrated for the region (Figure)

Figure 8.40. Aerial photograph of Cesenatico

Nowadays no saltmarshes, seagrasses, mussel beds, oyster beds nor Sabellaria reef are present in Cesenatico

The only natural habitats remaining are some scattered vegetated patches of limited naturalistic value (Figure and Figure).

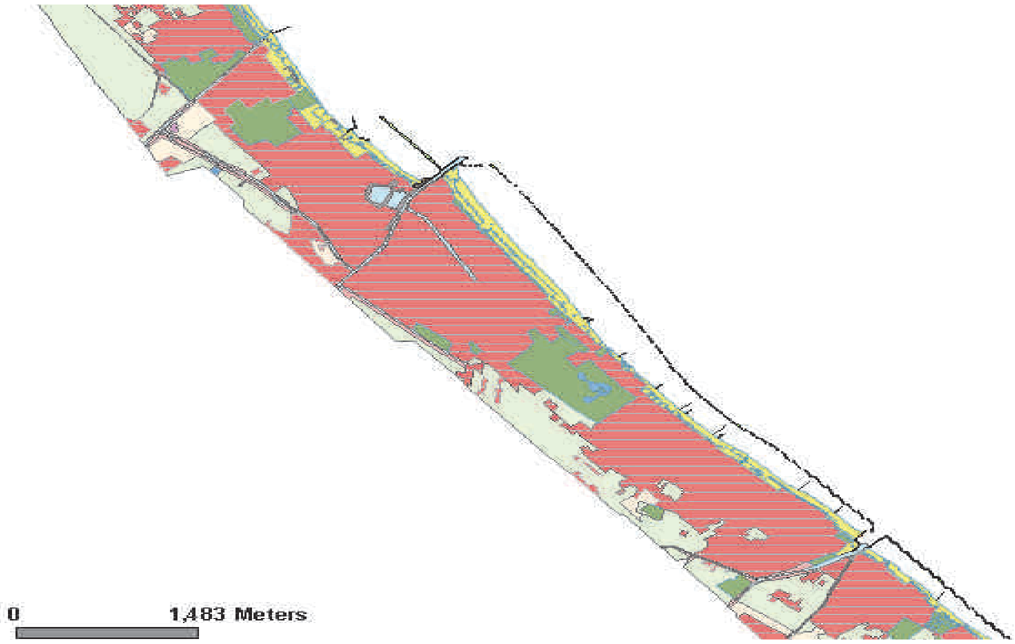


Figure 8.41. Distribution of vegetated areas (green) beaches (yellow) and coastal defence structures in Cesenatico (from Sistema informativo del mare e della costa (website of geological service of Emilia Romagna Region)

Figure 8.42. Appearance of the scattered vegetated patches and of the beach systems (with the artificial dune created seasonally to protect the bathing facilities) in Cesenatico

The seabed is gently sloping (about 6 m/Km), and a wide surf zone is exposed to waves raised by prevailing south-easterly and north-easterly winds in summer and winter, respectively. Shallow subtidal bottom sediments consist of fine to medium sands, and are colonised by assemblages generally dominated by bivalves (

Table).

Table 8.9. List of dominant species found in macrobenthic samples in shallow sandy bottoms nearby Cesenatico

Group	**Taxon**
Gastropods	*Cyclope neritea*
	Nassarius nitidus
Bivalves	*Scapharca inaequivalvis*
	Cerastoderma glaucum
	Mactra stultorum
	Tellina tenuis
	Donax semistriatus
	Chamelea gallina
	Tapes philippinarum
	Solen marginatus
	Lentidium mediterraneum
Crustaceans	*Ampelisca diadema*
	Bathyporeia guilliamsoniana
	Microprotopus maculatus
	Iphinoe sp. 1
	Iphinoe sp. 2
Polichaetes	Orbinidae sp.1
	Orbinidae sp.2
	Capitella capitata
	Capitomastus minimus
	Heteromastus filiformis
	Glycera tridactyla
	Glycera rouxii
	Nephtys sp.1
	Onuphis eremita
	Scolelepis squamata
	Spio decoratus
	Prionospio caspersi
	Owenia fusiformis
	Sigalion mathildae
	Policheata sp.1

Figure 8.43. View of shallow water habitats in Cesenatico

The whole coastline in Cesenatico is protected by an uninterrupted sequence of hard defence structures except for a small unprotected gap in the northern area. These breakwaters are colonized by assemblages typical of artificial structures in the region (Table 8.10)

Table 8.10. List of dominant species on coastal defence structures in the study region

Mytilus galloprovincialis
Crassostrea gigas
Ulva spp.
Sabellaria alveolata
Patella caerulea
Codium fragile ssp *tomentosoides*
Filamentous algae
Styela plicata
Didemnum granulosum
Antithamnion cruciatum
Bryopsis spp.
Gelidium spinosum
Monodonta mutabilis
Scytosiphon lomentaria
Encrusting algae
microfilm
Gracilaria verrucosa
Chthamalus spp.
Balanus spp.
Bryopsis plumosa

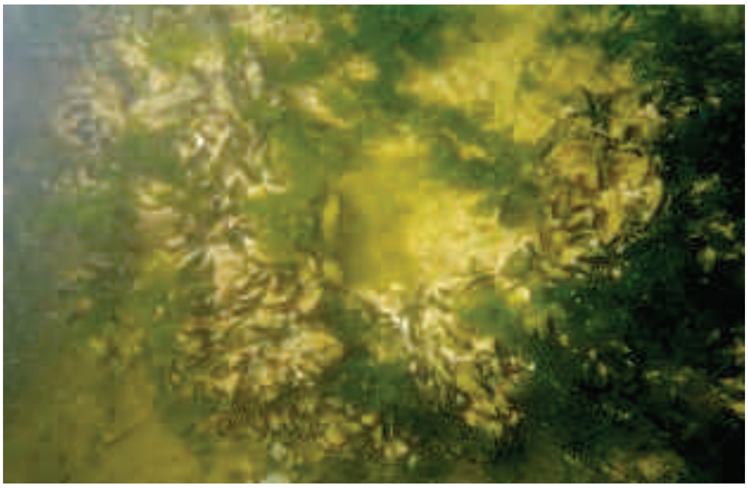


Figure 8.44. View of assemblages on coastal defence structures in Cesenatico

BELLOCCHIO

The study site Valli di Bellocchio is part of the Park of Po Delta River. The park of Po Delta river covers a very important part of the territory of the Emilia- Romagna Region: in the North from the Po di Goro, it develops including the whole historical delta of the river Po and the mouths of some Apennine rivers like the river Reno, the river Lamone, the Fiumi Uniti, the Bevano, and brackish wetlands situated along the Adriatic coast and the near hinterland like the Sacca di Goro, the Commacchio Valleys, the "Piallasse" of Ravenna, the Salines of Cervia, the inner wetlands of the Campotto Valleys, woods and pinewoods like the Bosco della Mesola and the Pinewoods of San Vitale.

Figure 8.45. Map with location of Valli di Bellocchio

Valli di Bellocchio site is situated between SS Romea road and the sea (Fig. 7 and 8) , within the provinces of Ferrara and Ravenna. The site is classified and protected as a nature reserve and has been proposed as "Sites of community importance" for the EU network Natura 2000.

Figure 8.46. Aerial photograph of Valli di Bellocchio

The protected area is made up of three distinct areas referred to as Bellocchio I, II and III. It is a brackish lagoon formed by a clay plateau with a large landfill on the left bank of the river Reno. The area is subject to cyclical flooding of the river Reno with their contribution of sediment and marine ingression dependent on tides and storms.

Sacca di Bellocchio I was established as the zoological and bioenergetics nature reserve in 1972, it occupies an area of 163 and is located in the province of Ravenna. Sacca di Bellocchio II was established as a nature reserve in 1979. It occupies an area of 83 ha in the provinces of Ferrara and Ravenna. Sacca di Bellocchio III was established as a nature reserve in 1981 and biogenetic. It occupies an area of 38.30 and it is located in the province of Ravenna.

Figure 8.47. Photograph of Valli di Bellocchio

As for the flora, the following habitats of Community importance within the meaning of Annex I to Directive 92/43/EEC are reported:

- Coastal lagoons

- Mediterranean salt steppes (***Limonietalia***)

- Mediterranean salt meadows (***Juncetalia Maritima***)

- Dune with forests of *Pinus pinea* and/or *Pinus pinaster*

- *Spartina* meadows (***Spartinion maritimae***)

- Estuaries

- Embryonic shifting dunes

- Shifting dunes of the beach ridges with the presence of *Ammophila arenaria* ("white dunes")

- Mediterranean humid grasslands of tall-herb-***Molinio Holoschoenion***

- Coastal dunes with *Juniperus* spp.

- Fixed coastal dunes with herbaceous vegetation ("gray dunes")

- Annual vegetation pioneer *Salicornia* (*Salicornia veneta* to report) and other species of mud and sand

- Annual vegetation of drift lines

- Dune grasslands with ***Malcolmietalia***

A relatively recent surveyed of the vegetal communities of the littoral saltmarsh Sacca di Bellocchio (Ravenna and Ferrara provinces), analyzed and classified following the Interpretation Manual of European has allowed to recognize: 1150*- Coastal lagoons; 1210 - Annual vegetation of drift lines; 1310 - Salicornia and other annuals colonising mud and sand; 1320 - Spartina swards (Spartinion maritimae); 1410 - Mediterranean salt meadows (Juncetalia maritimi); 1420 - Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi); 2120 - Shifting dunes along the shoreline with Ammophila arenaria (white dunes) (Merloni, 2007). Below a syntaxonomical scheme of the detected communities is reported (Merloni, 2007)

- RUPPIETEA MARITIMAE J.Tx 1960
    - Ruppietalia maritimae J.Tx. 1960
        - *Ruppion maritimae* Br.-Bl em. Den Hartog et Segal 1964
            - *RUPPIETUM CIRRHOSAE* Iversen 1941
- THERO-SALICORNIETEA Pign. 1953 em. R.TX. 1974
    - Thero-Salicornietalia Pign. 1953 em. R.TX. 1974
        - *Salicornion patulae* Géhu et Géhu-Franck 1984
            - *SALICORNIETUM VENETAE* Pign 1966
            - *SUAEDO MARITIMAE-SALICORNIETUM PATULAE* (Brullo et Furnari 1976) Géhu et Géhu- Franck 1984
- CAKILETEA MARITIMAE R.Tx. et Prsg. 1950
    - Euphorbietalia peplis R.Tx. 1950
        - *Euphorbion peplis* R.Tx. 1950
            - *SALSOLO KALI-CAKILETUM MARITIMAE* Costa et Manz. 1981 corr. Riv.-Mart. et al. 1992
    - *Thero-Suaedion* BR.-BL. 1931
        - *SALSOLETUM SODAE* Pign. 1953
        - *SUAEDO MARITIMAE-BASSIETUM HIRSUTAE* Br.-Bl. 1928
- SPARTINETEA MARITIMAE (R.Tx. 1961) Beeft., Géhu, Ohba et R.Tx. 1971
    - Spartinetalia maritimae (R.Tx. 1961) Beeft., Géhu, Ohba et R.Tx. 1971
        - *Spartinion maritimae* (R.TX. 1961) Beeft., Géhu, Ohba et R.Tx. 1971
            - *LIMONIO NARBONENSIS-SPARTINETUM MARITIMAE* (Pign. 1966) Beeft. et Géhu 1973
- EUPHORBIO-AMMOPHILETEA ARUNDINACEAE J.M. et J. Géhu 1988
    - Ammophiletalia arundinaceae BR.-BL. (1931) 1933 em. J.M. et J. Géhu 1988
        - *Ammophilion arundinaceae* BR.-BL. (1931) 1933 em. J.M. et J. Géhu 1988

ECHINOPHORO SPINOSAE-AMMOPHILETUM ARUNDINACEAE Géhu, Riv.-Mart., R.Tx. 1972 in Géhu et al. 1984

SARCOCORNIETEA FRUTICOSAE R.Tx. et Oberd. 1958

- Sarcocornietalia fruticosae (Br.-Bl. 1931) R.Tx. et Oberd. 1958
  - *Sarcocornion fruticosae* Br.-Bl. 1931
    - *PUCCINELLIO FESTUCIFORMIS-SARCOCORNIETUM FRUTICOSAE* (Br.-Bl. (1928) 1952) Géhu 1976
    - *PUCCINELLIO CONVOLUTAE-ARTHROCNEMETUM MACROSTACHYI* (Br.-Bl. 1928) Géhu ex Géhu et al. 1984
    - *PUCCINELLIO FESTUCIFORMIS-HALIMIONETUM PORTULACOIDIS* Géhu, Biondi, Géhu-Franck et Costa 1992
    - *HALOCNEMETUM STROBILACEI* Oberd. 1952 em. Géhu 1994
    - *SARCOCORNIETUM DEFLEXAE* (Br.-Bl. 1931) Lahondère, Géhu et Paradis 1992

JUNCETEA MARITIMI Br.-Bl. 1952 em. Beeft. 1965

- Juncetalia maritimi Br.-Bl. 1931
  - *Puccinellion festuciformis* Géhu et Scopp. 1984 in Géhu et al. 1984
    - *LIMONIO NARBONENSIS-PUCCINELLIETUM FESTUCIFORMIS* (Pign. 1966) Géhu et Scopp. 1984 in Géhu et al. 1984
  - *Juncion maritimi* Br.-Bl. 1931
    - *PUCCINELLIO FESTUCIFORMIS-JUNCETUM MARITIMI* (Pign. 1966) Géhu et al. 1984
    - *JUNCETUM MARITIMO-ACUTI* Horvatic 1934
  - *Elytrigio athericae-Artemision coerulescentis* (Pign.1953) Géhu et Scopp. 1984, corr. Pirone 1995
    - *ELYMETUM ATHERICI* Pellizzari, Merloni et Piccoli 1998

RHAMNO-PRUNETEA Riv.-God. et Borja Carbonell 1961

- Prunetalia spinosae R.TX. 1952

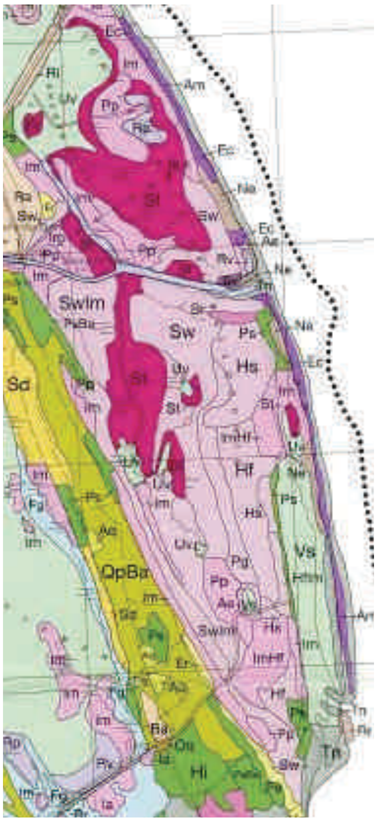


Figure 8.48. Vegetation map of Valli di Belloccio

As for the Fauna, most birds are reported, among which the following species included to the Annex I to Directive 79/409/EEC are reported:

Gavia stellata; Gavia arctica; Podiceps auritus; Phalacrocorax pygmeus; Botaurus stellaris; Ixobrychus minutes; Nycticorax nycticorax; Ardeola ralloides; Egretta garzetta; Egretta alba; Ardea purpurea; Ciconia nigra; Ciconia ciconia; Plegadis falcinellus; Platalea leucorodia; Phoenicopterus ruber; Pernis apivorus; Circus aeruginosus; Circus cyaneus; Circus pygargus; Pandion haliaetus; Porzana porzana; Porzana parva; Grus grus; Himantopus himantopus; Recurvirostra avosetta; Pluvialis apricaria; Philomachus pugnax; Gallinago media; Limosa lapponica; Tringa glareola; Larus melanocephalus; Larus genei; Gelochelidon nilotica; Sterna sandvicensis; Sterna hirundo; Sterna albifrons; Chlidonias hybridus; Chlidonias niger; Asio flammeus; Caprimulgus europaeus; Alcedo atthis; Acrocephalus melanopogon